Liberté • Égalité • Fraternité
RÉPUBLIQUE FRANÇAISE
PREMIER MINISTRE

Centre d'analyse stratégique

Avril 2009

# An economic approach to biodiversity and ecosystems services

## Contribution to public decision-making


Bernard Chevassus-au-Louis, chairman of the workgroup
Jean-Michel Salles, vice-chair
Jean-Luc Pujol, general rapporteur


Rapports et documents

Liberté • Égalité • Fraternité
RÉPUBLIQUE FRANÇAISE

PREMIER MINISTRE

Centre d'analyse stratégique

# An economic approach to biodiversity and ecosystems services

## Contribution to public decision-making


**Bernard Chevassus-au-Louis**, chairman of the workgroup

**Jean-Michel Salles**, vice-chair

**Jean-Luc Pujol**, general rapporteur

*Main authors:*

Bernard Chevassus-au-Louis
Jean-Michel Salles
Sabine Bielsa
Dominique Richard
Gilles Martin
Jean-Luc Pujol


April 2009

# Contents

*(See the detailed table of contents at the end of this work)*

## Summary

This report has two main objectives:

- to present and critically analyse the methods that can be used to estimate the economic values of biodiversity and ecosystem services,
- to apply these methods to the ecosystems present in France in order to provide reference values that can be used in the socio-economic evaluation of public investments, in particular.

After briefly mentioning the discussions initiated at the start of the 1980s on the subject of sustainable development and the links between biodiversity and human activities, the report goes into details (chapter 2) about the major socio-economic issues that biodiversity and ecosystem services represent for France, not only at present but also in the future. It presents possible courses of action for integrating the economic aspect into the approach to biodiversity.

Chapter III analyses how the law has gradually got to grips with these problems of the economic value of biodiversity and how it has handled them. One of the major messages is that setting a value does not imply the opening of a free market and that any trade-offs must always be regulated.

The concepts related to the notions of biodiversity, ecosystem services and the links between these two notions are then specified along with the different indicators that can be used to describe the state of biodiversity, its evolution and the pressures weighing upon it. Chapter 4 insists on the fact that, even though biodiversity is a complex subject with many aspects, we can now make a judgement about its state and its evolution in a given place. Finally, the hypotheses selected by the group for linking the biological concepts and economic analysis are presented, in particular, the option of distinguishing between "remarkable" biodiversity and "general" or "ordinary" biodiversity and approaching the economic evaluation of these two categories in a different manner.

Chapter V presents the theoretical and methodological frameworks that can be used to evaluate biodiversity and services linked to ecosystems that are not subject of direct economic exchanges (some of them are). We emphasise the fact that, although use values can be approached with relatively robust methods, notably on the basis of costs or effects on productivity, non-use values are often important, in particular for remarkable biodiversity, but their estimation is still much more uncertain. A review of the published results leads us to validate the evaluation of biodiversity and ecosystems by means of the services linked to them.

Details about the main research issues identified by the group are given in chapter 6. This highlights the importance of setting up sustainable biodiversity observatories that also take into account the monitoring of human activities and the pressures that they exert and on the need for concrete and multidisciplinary work to evaluate certain ecological services in a spatialized manner, especially those of protection and regulation.

Chapter VII deals in detail with the technical aspects of drawing up reference values and emphasises that there are currently important differences between the question of biodiversity and that of fixing carbon, often mentioned as a reference. After having demonstrated the limits of economic analysis of remarkable biodiversity, this chapter deals with several specific cases related to ordinary biodiversity, in particular that of temperate forests, and examines the way in which the use of these values may influence changes in land use. It sets out the questions of a procedural nature that arise, both in the drawing up and the use of reference values, in order for this

approach to be considered acceptable by the stakeholders. Finally, this chapter presents the non-monetary approaches that can be used, especially in the compensation practices.

General conclusions examine the main responses made to the referral and propose some short-term recommendations for the use and continuation of this work.

## Operational synthesis concerning the drawing up of reference values

This synthesis deals with the main subject of a report, i.e. the procedure followed in order to draw up the first reference values for France's biodiversity and ecosystem services. In practice, in order to propose these values, combining current ecological, legal and socio-economic knowledge, **the workgroup adopted several options that should be highlighted**, both because they may be disputed and because they condition the values proposed. **If these options are validated, they could be used as a plan for creating a first methodological guide to drawing up reference values**

1. The first option was to prioritise *ex-ante* socio-economic calculation, i.e. to provide estimates that are as reliable as possible **of all of the losses that may result from the altering of an ecosystem** and that must be endured (or compensated for) by society. Therefore, the group considered that other related questions, such as any claims from economic players for remuneration in order to prevent all or part of these losses, or other uses of this reference system, should not have a retroactive effect on the drawing up of these estimates.

2. Looking at the available data, and also at the fact that the objective of "halting the decline in biodiversity by 2010" was not yet sufficiently specific to consider a cost/effectiveness analysis, **the group decided to draw up its reference values using a cost/benefits approach.** It nevertheless asked questions about the effectiveness of this reference system, i.e. its ability to encourage reconsideration of changes in land use, in particular the destruction of areas with permanent vegetation cover (forest and pastures).

3. Given the complexity of the notion of biodiversity, the workgroup has proposed to **distinguish between two categories:**
    - **one, said to be "remarkable"**, corresponding to entities (genes, species, habitats, landscapes) that society has identified as having an intrinsic value and based mainly on values other than economic,
    - **the other, said to be "general"** (or "**ordinary**"), without any intrinsic value identified as such but which, by the abundance of its entities and the multiple interactions between them, contributes in various degrees to the functioning of ecosystems and the production of the services that our societies find in them.

   We should emphasise that **the distinction of "remarkable" entities is not purely biological:** it combines ecological (rareness or determining functional role in the case of species), sociological (the "heritage" aspect), economic (the predominance of the non-use values over the use values) and in some case legal (areas protected by statute, species on an official list) criteria.

4. In the fourth option, linked to the previous one, the workgroup proposed that, even though it analysed the economic evaluations of **remarkable biodiversity, it would only use those evaluations in a subsidiary manner in the discussions about the conservation of those entities.** In other words, the workgroup considered that today it is neither credible - in terms of reliability and relevance of estimates - nor appropriate - in terms of inclusion in discussions involving many values - to propose reference values for remarkable biodiversity.

5. In the case of **general biodiversity**, the workgroup opted **not to try to evaluate it directly but to do it on the basis of the ecosystem services that benefit society.** The report argues for an underlying hypothesis concerning a relationship of proportionality between the fluctuations in biodiversity and the extent of these services. This option is based, in particular, on the fact that, contrary to remarkable biodiversity, general biodiversity is currently perceived in an imprecise way by people and this lack of perception limits the relevance of direct estimation methods based on the declaration of preferences.

6. **The group relied on the classification proposed by the *Millennium Ecosystem Assessment (MEA)* for evaluating these services**.
   This classification distinguishes between four sets: "**support services**" that cannot be directly used by humans but condition the correct working of ecosystems (recycling of nutrients, primary production), "**provisioning services**" that result in goods that can be used by humans (food, materials and fibres, freshwater, bioenergy), "**regulating services**" meaning the ability to modulate phenomena such as the climate, the occurrence and scale of diseases and different aspects of the water cycle (high and low water levels, physical and chemical quality) in a sense favourable to humans and, finally, "**cultural services**" meaning the use of ecosystems for recreational, aesthetic and spiritual purposes. **In order to avoid double accounting, in particular, the group followed the MEA's recommendation and did not evaluate the support services**, because they considered that in reality they condition the continued existence of three other sets of services: therefore, like biodiversity, they will be evaluated by means of those services.

7. Linked to the first point, **the group decided to consider not only "dynamic" services, i.e. in terms of flows** (carbon fixing, water production, tourist visits, etc.), **but also "static" services** (stability of soils, retaining a stock of carbon). It considered that in practice the value of the potential loss of these services if the ecosystem was destroyed (increasing erosion, quicker or less quick release of $CO_2$) should be deducted from the socio-economic accounts of an operation that would lead to that destruction, which is the same thing as marking up the loss avoided to the credit of those ecosystems.

8. Amongst the different components of the notion of total economic value, the group prioritised the use values (in the widest sense, i.e. including the potential uses in a more or less long-term). In practice, it considered that, as for remarkable biodiversity, the methods for estimating non-use values were less robust and their legitimacy was more open to dispute. Similarly, **the group prioritised methods considered to be robust for estimating these use values** (prices recorded, actual expenditure, restoration or replacement costs).

9. Given the complex problems of value transfer, **the group limited itself to the services for which it had French studies (or those from countries that are ecologically or socio-economically close)** and which, in addition, provide a relatively homogeneous reference system (or whose heterogeneous nature can easily be explained). The result is that **certain services** (for example, the effects on health or protection against natural disasters), **for which the references were limited, inconsistent or very exotic, have not been evaluated, even if it is legitimate to assume that they have a high value.**

10. The group recognised that **there was no advantage in using any other discounting rate than that used for other aspects of socio-economic calculation** (4% today, decreasing after 30 years) for taking the long term into account. Conversely, it proposed that an **average increase of about 1% (or more in situations of irreplaceable losses) in the relative prices of** ecosystem services in relation to manufactured goods should be retained until 2050. This means that annual value of the service must be multiplied by 40 to obtain the total discounted value. Dynamic models of the evolution of biodiversity, integrating the scientific strategy for evaluating it, would have to be perfected in order to arrive at more accurate estimates of this increase in relative prices and its change over time that are adapted to the situations encountered.

By applying this procedure, the group has arrived at some reference values, especially that of the average value to be ascribed to mainland forest ecosystems, 950 Euros per hectare per year (giving around 35,000 Euros per hectare in total discounted value), with a range of variation of from 500 to 2,000 Euros per hectare per year depending on, in particular, the recreational use or tourist visits and the method of managing the ecosystem. A minimum value of around 600 Euros is also proposed for pastures used in an extensive manner.

In conclusion, the group emphasises that it does not propose reference values for the whole of biodiversity but **only for the use values of ecosystem services related to general biodiversity that can currently be given monetary values in a way that it considers robust.** But, due to this, the group considers that **the estimates proposed are *minimum* estimates, which can therefore incontestably immediately replace the null value used for biodiversity in socio-economic calculations.**

At the same time, the group calls for complementary work that takes other services into account and that may significantly increase these values.
To refine its work in the short term, the group particularly recommends that:

- **this critical synthesis work should be extended rapidly** to all of the national ecosystems for which data coming from similar ecosystems is available, based, where appropriate, on the work in progress for an MEA France,
- the **average values are spatialized** at least to a departmental scale, to take into account both ecological and socio-economic particularities,
- not only the current value but what is called the "maximum plausible value" in the medium term (30-50 years) is defined for this spatialized data, integrating, in particular, the predictable variations in the rates of use of the different ecosystem services,

- **the structures** responsible for creating this reference system, getting it discussed, regularly updating it and concretely using it **should be specified, determining the procedural methods that endow it with an area of application and legitimacy that lead to its adoption by all of the operators concerned.**

## Analytic summary of the chapters

The workgroup was entrusted with a mission that identified four major questions:

- *"Draw up an assessment of scientific knowledge on the theme of ascribing monetary values to the services provided by ecosystems and the value of biodiversity.*
- *Analyse the socio-economic issues of biological diversity in France, including the Overseas Departments and Territories.*
- *Propose specifications for any later research.*
- *Estimate the first reference values for taking biodiversity into account that could be used in the socio-economic studies related to infrastructure projects".*

## The framework: the *ex-ante* microeconomic approach

**The first chapter of the report resituates these questions in the international, European and French research initiated at the end of the 1980s on the theme of sustainable development.** This research led to the perception of the strong links between biodiversity and human activity and hence to the posing of questions about the economic evaluation of biodiversity and the services attached to it.

The workgroup decided to frame its discussions on the basis of two distinctions:

- that between macroeconomic concerns aimed at global evaluations (relations between economic development and biodiversity and ecosystem services. National compatibility) and microeconomic approaches analysing the impacts of choices and behaviours on an *a priori* more local scale (infrastructures, pollutions) whose effects we wish to measure,
- that between the *ex-ante* approaches (support for public decision making, via the socio-economic evaluation of projects, in particular) and the *ex-post* approaches (repairing damage, ecological compensation).

In the spirit of its mission statement, **the workgroup prioritised the focusing of its work on ex*-ante* micro economic evaluation, providing reference values for socio-economic calculations.** In reality, whilst other things (time values, noise, pollution, etc.) are already taken into account in the cost/benefits analysis of public choices, the effects on biodiversity are currently still considered *de facto* to be null.

Whilst choosing this option, **the European and national objective of halting the loss of biodiversity by 2010 – an objective that implies that any loss of biodiversity in a given place must be at least compensated for by a gain in biodiversity in another place in the country – has also been taken into account and its consequences analysed.** In practice, it implies that an economic evaluation of biodiversity must come within a system of so-called "strong sustainability", i.e. **it**

**cannot be used to support transactions with other elements of well-being that may compensate for losses of biodiversity.**

Finally, **the workgroup also thought it useful to briefly examine some *ex-post,* approaches, in particular that of compensation**, to identify the specific features in relation to the *ex-ante* approaches.

## The socio-economic issues of biodiversity

**In response to the second question in the referral, chapter 2 presents the different aspects of the socio-economic issues of biodiversity and ecosystem services.** It places the French context, including Overseas France, within the wider framework of the issues of worldwide biodiversity.

**The first issue is that of a change in our perception** which will condition the priorities of the actions to be implemented: whereas public perception of biodiversity is often limited to a few symbolic plant and animal species, it is essential that biodiversity is viewed from the angles of its omnipresence as the basis of life and its multiple interactions with human societies, whether it be as the reservoir of food and medicines, or the support for the great biogeochemical processes, the chemical industry or even creative inspiration. Therefore, we put the accent on multiple goods and services which benefit our human societies, emphasising a few examples of emerging issues and insisting on the fact that the services will no doubt be even more important in the future than they are today.

**The second issue is related to the description and understanding of the dynamics of biodiversity.** At the dawn of a possible sixth extinction of biodiversity, analysis of the socio-economic issues of biodiversity is inseparable from an evaluation of its state and evolution under the effects of increasing pressures. In addition, France bears major responsibility in this field because it hosts a notable part of worldwide biodiversity in its overseas territories. Yet our knowledge of biodiversity, its relationship with functions and services that human societies derive from it and processes that govern its evolution is largely lacking, which makes evaluations for decision-making support difficult and complex. These uncertainties become particularly acute in the light of climate change which calls into question the ability of ecosystems to adapt and above all continue to provide the services on which we depend.

**The third issue is that of the mobilisation of players.** This chapter summarises the nature of international, European and French commitments that have already been made in favour of biodiversity and the different operational tools already in place, whilst at the same time highlighting the variety of the players involved and the diversity of the issues - ethical, social and geo-economic - to be taken into account.

**The different initiatives taken are then presented along with the avenues for action for integrating the economic aspects into the approach to biodiversity:** "macro" type procedure - like the current initiatives on "The Economics of Ecosystems and Biodiversity" (TEEB), the revision of the United Nations system for national accounting, the "Initiative for a green economy"- or more targeted – like the changes in the tax system, the re-examination of public subsidies with negative effects on biodiversity, the payment mechanism for ecosystem services and the integration of biodiversity into company accounts.

## The legal approach to the value of biodiversity

**and programmes, a specific chapter is dedicated to the law's contribution to this discussion about this ascribing of monetary values to biodiversity and the services provided by ecosystems, particularly in the light of recent changes in the environment law, especially the European Union law.** Positive law has in practice generated new needs concerning the "value of biodiversity", both by means of directive 2004/35/CE concerning environmental responsibility and by means of the numerous laws that decree that negative impacts must be prevented or if not reduced and, as a last resort, compensation made for the residual impacts.

Starting from an analysis of the judgements handed down in a matter of repairing damage caused to the environment and the rare statutes, particularly repressive, that convey social reactions to attacks on biodiversity, the arguments in this chapter ask whether the law can admit the principle of giving a value to biodiversity, particularly in the socio-economic studies related to infrastructure projects. They point out that the Council of State has endorsed the "cost/benefits" reporting method in the preparation for public decision making and that therefore there is nothing shocking about attempting to give biodiversity a monetary value in socio-economic calculations – but that this ascribing of a value to biodiversity does not have the effect of making it legally marketable good.

Beyond this, and in so far as the laws favour recourse to compensation mechanisms, both for repairing damage already caused by infrastructure projects and for anticipating that which may be, the chapter puts the accent on **the need to set up independent regulatory institutions and mechanisms that are as objective as possible.** This independence both from the government and the operators in the sector must be proven. It is important to prevent confusion arising between the exercise of state power (to prohibit or authorise, implement administration policies and impose obedience with prescriptions) and the determination of the conditions of exchange and compensation, so that the administrative decision is not perverted by the inversion of the "prevent-reduce-compensate" triad. It is equally important to prevent the operators in the sector from capturing the exchange and compensation mechanisms and to guarantee the transparent operation of these mechanisms.

Finally, the setting up of an independent authority, necessarily very specialised and operating within a framework strictly defined by the laws, cannot absolve political power from its final responsibility for decisions.

## The description of biodiversity: concepts and biological indicators

**Firstly, chapter IV presents the main biological concepts underlying notions of biodiversity and ecosystem services and the indicators that are currently available or could be considered for describing them.**

The notion of biodiversity has undergone important changes in its development from an area of interest solely to biologists and protectors of nature to one that concerns politicians: the static view centred on species that prevailed at the start of the 19th century has gradually been replaced by a changing and functional view including diversity within species (in particular the diversity of genes), the diversity of associations of species populating the ecosystems ("ecological" diversity) and above all the importance of the interactions between all of these components.

Furthermore, the "new frontier" notion introduced due to the intensive exploration of new environments (ocean depths, tropical forests), has marked the end of the 20th century: it highlights the unsuspecting breadth of species diversity- mainly still to be described - but also the fact that **our perception of biodiversity is still based today on species that are relatively large and easily observable but which are the exception rather than the rule among living organisms.**

In the case of the indicators, the first proposals for global quantitative measurements date back to the 1950s and correspond to the scientific objectives of synthetic and comparative descriptions of populations. Since the 1960s, the perception of accelerated reduction in biodiversity has motivated the efforts to build simple indicators that can be understood by political decision-makers and more widely by the general public. The objective was to be able to monitor, on larger or smaller scales (ecosystems, landscapes, ecoregions), the variations in biodiversity over time (indicators of state) and also indicators of the pressure (or interaction) of human activities on biodiversity.

Ideally, these indicators should take into account the number of different entities present (richness), their relative abundance (divergence from equi-distribution) and the smaller or greater diversity of those entities (evolutionary or functional differentiation). In addition, they should report on the absolute abundance of species or populations and the spatial organisation of the entities that make up ecosystems. **Several arguments make the case - and we strongly emphasise this point- in favour of the monitoring of biodiversity based on an estimation of the variations in abundance of species:** these variations integrate the different mechanisms that govern their future, they are quicker and more continuous than the variations in specific diversity and, finally, a whole variety of specific indicators can be derived from them (dynamics of species, ecological services, quantification of pressures, responses, etc.)

Today, numerous indicators are available at world level and are presented in this chapter. **Most of them do not benefit from systematic organisation of the collection and analysis of data coordinated at world level.** However, in Europe the SEBI 2010 programme, run by the European Environment Agency (EEA), aims to organise, in liaison with the countries that are members of its network, the regular documentation and enrichment of a set of 26 pertinent indicators for the analysis of biodiversity; France has taken inspiration from these indicators when defining its own set of monitoring indicators for the National Biodiversity Strategy. Discussions are taking place in the EEA on developing an indicator related to the integrity of ecosystems.

Thus, these biodiversity indicators must give an account of a "multi-dimensional" subject not only by its different levels of organisation (genetic, specific and ecological diversity) and by the diversity of the entities within each of those levels but also by the complexity of its perceptions (by ecologists, taxonomists, naturalists, managers, economists). Therefore, the quantification of biodiversity is a particularly ambitious objective and **the wish to define a single indicator that accounts for all of the**

**aspects of biodiversity is an illusion. Conversely, the state of biodiversity in a given place in relation to a particular concern can be characterised using an array of relevant indicators.**

**However, within this context of still imperfect data and tools, some variations in biodiversity can only be detected indirectly, by means of any variations in the ecosystem services to which that biodiversity contributes.** Significantly, the most numerous and most successful economic studies are based much more on the indirect evaluation of ecosystem services than on biodiversity. Due to this, **the workgroup chose four options for relating the biological approaches to biodiversity and the economic analysis:**

1. **to distinguish the "heritage" aspect within the biodiversity of a given area** (or its "remarkable" aspect, i.e. the existence of entities of special interest) **from its "general" or "functional" aspect** (linked to the interactions between more or less abundant "ordinary" entities that contribute to the production of ecosystem services) **and treat these two aspects in a differentiate way,**

2. **to assert from the outset that the economic analysis of "remarkable" biodiversity must be a "subsidiary" element in relation to the multiple criteria (ecological, ethical, cultural, aesthetic) to be taken into account.** This option is based on the limits of the methods of economic evaluation that can be used, in particular, and on the fact that the very possibility of "substitution" with other goods seems to be excluded *a priori,*

3. **to approach the economic analysis of "ordinary" biodiversity by means of the ecosystem services to which it contributes and not in a direct way,**

4. finally, **to work using a "median" hypothesis of a positive linear relationship between general biodiversity and ecosystem services.**
   This option is based on the principle that the economic evaluation of the reduction of these services will provide a relevant measurement of the associated losses of biodiversity. It enables us to consider that the observed or potential variation in indicators of biodiversity in a given environment could be "given a monetary value" by assuming a relatively similar variation in the ecosystem services of that environment.

**As far as the distinction between "remarkable" and "ordinary" biodiversity is concerned**, though the group accepted the principle, it only sketched out the factors (ecological but also socio-economic) that may be taken into account to identify these "remarkable" entities.

## Economic evaluation: basics, methods and results

**Chapter V is dedicated to the critical presentation of the economic approaches that can be used and their main results.**

The evaluation of biodiversity and ecosystem services throws up a set of difficulties. Economic analysis offers a framework for resolving them that is sometimes problematic and whose relevance is often contested but which is likely to make evaluation consistent and comparable with the other social issues. This framework is characterised by an anthropocentric approach which evaluates the choices according to their consequences, measured in terms of variations in individual and social well-being. **Its principle is based on individual preferences, assumed to guide the choices of agents** (individuals, households or businesses) towards the search for the greatest well-being; which implies that the value that will be assigned to environmental assets is dependent on both the information known by the agents and their abilities to infer consequences from it. Thus, the problems of adding individual preferences confer some advantages on the monetary standard as a one-dimensional measurement of preferences (approach by means of an estimate of willingness to pay).

In the case of biodiversity and ecosystem services, the fact that most of the agents lack familiarity with these notions introduces greater inaccuracy into the results obtained by the use of individual or partially added preferences that we find in other fields. **This inaccuracy, which should diminish as people become increasingly aware of these questions, leads us to pay special attention to the notion of "merit goods" for which an evaluation resulting from the agents' preferences cannot be used directly to justify collective choices.** 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 services provided by the ecosystems, but also due to an incomplete or biased perception of that value, related to the indirect and only vaguely perceivable character of the services provided.

In spite of this limitation, economic evaluation provides a framework that can be used to integrate the many aspects of the value of ecosystems. Over and beyond the utility **gained from the direct use of these assets, a set of extensions has been used to formulate a very wide concept of value which tends to go beyond the strictly utilitarian framework:** the indirect uses to enable us to integrate the services that do not imply direct interaction between users and ecosystems (cases of uses deferred over time); the option and quasi-option values reflect the insurance related role of biodiversity and the gains linked to the improvement of information when faced with irreversible choices; the non-use values demonstrate the existence of ethical preferences whose integration into the evaluation seems both necessary (biodiversity's contribution to our well-being is obviously greater than the use values alone) and problematic, because measuring them is difficult and introduces heterogeneousness into the analysis framework.

We should remind ourselves in this respect that the values that we may obtain do not necessarily have an absolute character but are mainly used to take into account the aspects related to biodiversity in the classification of solutions between which the public decision maker must choose. When the assignment of such values and their use in traditional evaluation prove to be too uncertain, it is still generally possible to resort to other methods for making this classification.

**In part, the consequences of choices in the matter of conservation of biodiversity concern far-off time horizons, which gives particular importance to the options selected for taking future issues into account.** Therefore, the discounting question is a major issue. The general recommendations in this field (a rate of 4% in the short to medium term decreasing in the long-term) should *a priori* be applied to the choices involving biodiversity. The consequence of the hypotheses about the changes in relative prices (increasing for increasingly rare and natural resources and decreasing for manufactured goods benefiting from technical progress) should be a reduction in the apparent discounting rate of the value of these services (and therefore the weight of the preferences of the present generation on the freedom of choice of future generations). At the limit, in the case of irreplaceable assets that contribute to well-being, an implicit price increasing at the rate of the discounting rate should be attributed to biodiversity (Hotelling rule), thus attributing a potentially infinite value if the service has the potential to be provided indefinitely. **This argument contributed to the support for the proposal to treat the remarkable biodiversity in a distinct way using more complex methods that more accurately take into account the predictable duration of the associated service and the possibilities of evolution of these assets.**

On the basis of these concepts, several methods have been developed to elaborate the practical measurements of these values form the data deduced from the observation of behaviours on the direct or substitution markets or surveys aimed at collecting the preferences of agents faced with hypothetical scenarios. **However, the analysts are faced with a dilemma: only the techniques based on declared preferences can be used to take into account the values other than direct use but the results obtained by these methods are sensitive to their implementation conditions and sometimes difficult to interpret.** Nevertheless, the data gathered by the methods linked to observable behaviours alone (displacement costs, hedonistic prices) is not sufficient for guiding choices and that based on costs (of restoration, replacement, impact on the functions of production, agricultural in particular) comes up against the question of the effectively complete character of the substitution planned. Therefore, the creation and use of reference bases for formulating the values of transfers between ecosystems studied and ecosystems threatened, for which we only have superficial information, implies a great deal of discernment but seems to be a necessity.

In terms of concrete end results, we now have a large number of evaluations covering species, habitats and ecosystem services. But **many of these works cannot be used directly for drawing up reference values for integration into a traditional socio-economic evaluation.** Their creation was subject to multiple forms of bias (representativeness, level of information, etc.) and their results, usually expressed in terms of individual willingness to pay, are difficult to use for constructing global values for ecosystems different from those for which the results were obtained.

## The needs for research

**Chapter VI responds to the referral's third question about the needs for research, limiting itself to the most directly operational aspects.**

In the field of biological sciences, **the development of biodiversity databases, monitoring mechanisms and composite indicators, at different spatial scales over the whole of national territory seems to be the basis of any policy in this area.**
It also emphasises the importance of being able to integrate indicators of the pressure on biodiversity exercised by various human activities into these information systems. This concern once again brings up the question of defining the relevant spatial entities – known as "socio-ecosystems" – for describing, analysing and managing biodiversity. The current initiative of MEA France may well provide an answer to this question, on

condition that it takes care to take socio-economic data into account in the typology that will be established

Still in the field of biological sciences, the importance that the **notion of ecological equivalence** may take on in the future, particularly in compensation practices, is an incentive for developing procedures that are as explicit and transparent as possible for establishing that equivalence, including its margins of uncertainty, and setting up mechanisms for validating those procedures.

Finally, the emergence of the notion of ecosystem services and its use in economic analysis leads the workgroup to **specify the link between those services and the different aspects of biodiversity**, in particularly to define, for a given ecosystem, the modifications to biodiversity that are likely to have an impact, or not, on the extent of those services in the short, medium and long terms. In this area, experimental ecology approaches, integrating this long term aspect – approaches that have not been developed much in France – should be encouraged to complete the observation mechanisms.

**The challenges in the fields of economic and social sciences seem to lie much more in the lack of concrete work applying the available methods than in methodological developments.**
Therefore, specific incentives should be set up, in particular for the evaluation of protection and regulation services. In this case, it would undoubtedly be a good idea to promote coupled approaches, associating specialists in the physical environment, ecology, economics, law and risk management. The question arises not only for natural risks but also for human health, in its links with biodiversity and the environment, whether it is a matter of the modulation of the presence or the effects of pathogenic agents or polluting substances.

**Another important aspect of the mobilisation of the social sciences concerns the analysis of pertinent procedures for handling the sustainable management of biodiversity.** We mentioned it in the case of the drawing up of merit values and the regulation of their use, but also in the question of expressing preferences in the case of contingent evaluation. It can also be posed in the case of the reaction of a certain number of players in society to various incentive mechanisms that might be set up on the basis of the evaluation of ecosystem services: it would be naive to think that these players would not behave in a strategic way when faced with any regulatory or incentive based policy.

Finally, we mention the question of **taking the diversity of preferences vis-à-vis biodiversity into account**, whether it be a diversity of points of view between the different players present in an area or the tensions that can exist between the local assessments and those of other stakeholders, outside that area, who have different priorities. It would be simplistic at the very least to express this diversity in terms of an "average preference" and work on the interface between economics, sociology and political science could throw light on this problem.

**We end with the question of the law and the legal "status" of biodiversity.** The fact is that, in practice, only part of biodiversity (the genetic resources of domestic species, protected species, sites of special scientific interest, etc.) have a real status specifying the rights and obligations of public and private operators. On the contrary, ordinary biodiversity, including the herbaceous plants, land macrofauna and above all the soil and water micro-organisms, is considered to be part of the private property of those who own and use the land. Now that, as we have amply demonstrated, ordinary biodiversity seems to be a major determining factor in ecosystem services we might well ask legal science about the advantages of a change in its "status" and the rights that can be exercised over it. It is worth thinking about the idea of special goods law – just as there is a special contracts law. This discussion would undoubtedly involve an analysis of the ethical or philosophical basis of the status of nature and biodiversity in our post-industrial societies.

## Setting reference values

**On the basis of the methods and concepts presented in the preceding chapters, chapter VII examines the possibility of setting reference values for French ecosystems today, as well as the limitations of that exercise.**

After restating the definition of the reference value objectives, the first part introduces a few questions related to the drawing up of these values. Firstly, it stresses the importance **of defining a clear objective to be achieved, including in terms of indicators of success, timing and the geographical area concerned.** In this light, though the microeconomic objective of internalising the costs of the impacts on biodiversity is operational, the current macroeconomic objective of halting the decline of French biodiversity in 2010 does not seems to have been made sufficiently explicit. We then emphasise the limitations on the use of market prices (real or hypothetical) for establishing these values. Finally, we present and compare the advantages and the feasibility of cost/benefits and cost/effectiveness approaches in the case of biodiversity. We conclude that the data currently available is suited above all to input into a cost/benefits approach. Therefore, the values proposed will be based on the current uses and not on the cost of achieving a standardised protection objective faced with actions that constitute a threat biodiversity.

**The second part refers to recent work on the merit value of fixing $CO_2$ to show, in comparison, the main specific problems linked to biodiversity:** impossibility of defining a simple and unique indicator similar to the "ton of carbon", the often local character of impacts limiting the relevance of large scale "sources" and "sinks" audits, the very specific nature of local situations making the value transfers problematic, challenging of the very legitimacy and relevance of ascribing monetary values as well as the substitutable character of biodiversity. Finally, we emphasise that whereas the ton of carbon is both a "pressure variable", measuring all of the human influences on climate, and a "control variable", use to guide the corrective actions, biodiversity represents a "status variable" resulting from multiple pressures that should be identified and reduced by specific policies working on the control variables associated with those pressures.

Nevertheless, the dynamics of ecosystems must be taken into account in setting the values attributed to biodiversity in order to make pertinent comparisons, in the long term, of the consequences of the different options that may be proposed to a decision maker. Taking this into account accurately will presuppose the development of models linking the state of biodiversity (described by parameters yet to be defined) with control variables (in the sense of dynamic systems) reflecting the different human pressures.

In a third part, we deal with several methodological problems specific to the setting of reference values for biodiversity and ecosystem services:

- how to reduce the wide variation in estimates, which seems principally linked to an insufficiently accurate definition of the ecosystems studied?
- considered both in their present state and in their "path"? **We conclude that a typology of "socio-ecosystems" adapted to France, overseas France included, and a weighting introduced that takes into account not only their present uses but also their potential use in the medium term (30-50 years), by means of the notion of "maximum plausible value".**
- how can the values of an ecosystem's different ecological services or the services of a mosaic of ecosystems be added in an area? can we content ourselves with simply adding them up or must we introduce more complex weightings?
- can we limit ourselves to a metric relating the services to units of area (Euros per hectare and per year)? Certain services are not proportional to the area of ecosystems, the area to be considered is sometimes very different from the physical area modified and the location of the areas modified can, for an equal area, lead to very different impacts. **Consequently, setting a reference value by unit of area is only a first benchmark that must be refined on a case-by-case basis.** The example of the economic analysis of the "blue and green" network planned by the Grenelle Environment Forum is mentioned, in particular, because it shows the limits of this approach.

In a fourth part, we return to the proposed distinction between "remarkable" and "ordinary" biodiversity to demonstrate the limitations of economic analysis of remarkable biodiversity. The examples of symbolic animal species and plants of pharmaceutical interest underline the inaccuracy and lack of robustness of the estimates of monetary value proposed. **We conclude with the fact that, given the current state of our knowledge, the use of such values may make the discussions about the conservation of this remarkable biodiversity more complex rather than clarify them**.

We then deal with several concrete examples: the case of coral reefs, to demonstrate the importance of the ecosystem services of such environments and the legitimacy of heavy investments for protecting them, that of wetlands, to demonstrate the need for more specific studies of the French situation, then that of temperate forests, that seems to us to be a text-book case for illustrating the procedure to use in order to propose reference values for the different ecosystem services of these environments from a set of bibliographical data.

Based on the *Millennium Ecosystem Assessment* typology of services and analysing the role of biodiversity in the production of those services each time, this analysis leads us to propose an average reference value of around €970 ha/year for France for the set of forest services for which monetary evaluation with a certain robustness can be made (in fact, a minimum range of between 500 and 2,000 Euros depending on the real intensity of those services). **This value is significantly higher than those published up to now but this study demonstrates above all the need to better evaluate certain services (in particular, the services of regulation and protection against natural disasters) and strongly modulate this value according to the local** situations (the ecosystem's characteristics, management method, topography, population density, accessibility, etc.). We then give the outlines of a similar study in the case of permanent pastures, for which a reference value of around €600 ha/year could be proposed, if they are managed extensively.

Finally, we ask ourselves about whether such reference values are capable of influencing changes in land use, in particular the current tendency to reduce certain areas of permanent vegetation cover whose importance in terms of biodiversity is well known (meadows, fallow land). We show, in a first analysis, that the effective remuneration of the ecosystem services linked to these areas, in particular in the context of CAP subsidies, could actually correct, or even reverse, the differences in profitability between different types of production, in particular between annual crops and grassland stock raising. Still in this first analysis, we show that a single levy for the definitive loss of these services related to a development (for example, the creation of a tarmac car park) would be of the same magnitude as the costs of restoring similar ecosystems and would therefore enable us to finance such operations. Conversely, whether such levies based on the estimate of ecosystem service would be capable of influencing major overbuilding operations (urban development, transport infrastructures) seems more questionable, even if it can play a role in arbitration between various options that are more or less damaging to biodiversity. The question then posed is the limitations of an economic approach in relation to a regulatory approach protecting certain areas.

The rest of the chapter examines how, on the basis of the preceding general frameworks, these reference values could be concretely defined in the future and above all used when we set aside socio-economic calculations and consider remuneration, compensation and exchange mechanisms.

As far as the definition is concerned, it is a question of specifying the mechanisms to be set up to fix these values in a way that makes them effectively recognised and accepted as such by all of the players concerned. In particular, we insist on the importance of this "procedural legitimacy" in a context where the technical data is complex and significantly marked by uncertainty and where taking the long term into account will be as much a matter of political and ethical choices as of economic ones.

As far as the use of these values in other contexts than socio-economic calculations is concerned, the report discusses in particular the question of legitimacy and the appropriateness of the remuneration of private players involved in the production of ecosystem services. We show that if, on a theoretical level, this remuneration should be limited to solely the payment of the human capital invested, higher sums could be considered pragmatically in their cost/effectiveness approach.

Finally, the report presents a possible complementary approach, that of "service for service" ecological compensation. This approach was developed in the United States in the 1980s and is gradually being implemented in the European Union within the framework of the environmental responsibility directive. We point out the main differences between this approach and that of the monetary evaluation of services: no possible substitution with other elements of well being ("*no net loss*"), the possibility of establishing "equivalences in kind" without resorting to monetary values, a priority search for compensations that are nearby and affect ecosystems similar to the ecosystems affected. We conclude that if, as we propose, the geographic scale of

**management is limited these approaches could be more relevant and more operational - and perhaps provoke less conflict - than those based on monetary values. Therefore, it seems better, and this is also the option currently chosen, to reserve the approaches that ascribe monetary values mainly for *a priori* socio-economic calculations of the appropriateness of public investments and give priority to non-monetary approaches for handling the compensation for any residual impacts once the decision has been taken to make these investments.**

The question of regulation of transactions that may take place using such units of equivalence in kind is then discussed. We insist on the **need for an "independent authority"** that is involved in the key decisions (authorisation of the transaction, designation of the beneficiaries, validation of the monetary or other equivalences used). We also argue for the setting up of relatively local (regional or intraregional) exchange areas for these transactions even if the procedural roles have to be established at national or European level.

## Conclusions and recommendations

The general conclusions stress four points:

- When we mention the socio-economic issues today we often associates those of the reduction of biodiversity and those of climate change. This association is legitimate in so far as it is a question of strongly interconnected issues of equal importance. However, this should not mask the fact that the current decline in biodiversity is linked to many other factors, such as modification and fragmentation of habitats, the introduction of species and pollution, that have been at work for many years. Therefore, it is important that we control the damaging effects of these factors as quickly as possible in order to enable biodiversity to face up to the challenge of climate change and if possible moderate its extent and impacts.

Regarding the current state of knowledge and the needs for research, there are certainly methodological developments that need encouraging but we already have a wide enough range of biological indicators and economic approaches that we can mobilise and whose advantages and limitations we understand well enough. **This means that the main challenge is to obtain concrete data on the state of biodiversity and ecological services for the whole of the national territory at sufficiently precise spatial scales.** Furthermore, this data should be updated regularly and linked to the measurement of pressures that may affect these resources.

- In the case of certain ecosystems defined in quite broad manner, for example "temperate forests" or "permanent pastures", we can now propose more or less accurate estimates of the economic value of a certain number of ecological services associated with them. Transforming these different items of information into a global reference value, taking into account the inaccuracies, lacks (no valuation of certain services), local specific features, the long-term prospects for use of those services and the relative importance to give to different services, is an exercise that cannot be carried out on strictly technical and objective bases. **Therefore, this setting of reference values requires "deliberative procedures" whose methods and relevant territorial level are yet to be defined, but which must ensure, notably by adhering to criteria of**

**transparency and independence, the "social legitimacy" necessary for the practical use of such values.**

- We think that it would be a good idea to continue with this work quickly by creating critical syntheses of the data available for the other ecosystems in mainland France and the overseas departments and territories on the model of the approach followed for temperate forests, examining, in particular, the possibility of providing spatialized data, at departmental scale at least, on the economic value of different services.

Therefore, the workgroup recommends that:

1. This work of critical synthesis and spatialisation of data is rapidly extended taking into account future results from MEA France.

2. The permanent pluralist structure for setting and regularly updating the methodological frameworks and key parameters to be used by the operators responsible for drawing up the reference values is identified (or, if necessary, created).

3. Similarly, the decision-making places and processes responsible for applying these reference values to concrete operations and taking into account other aspects, in particular elements of remarkable biodiversity, are defined.

4. The national objective of halting the decline in biodiversity by 2010 is specified, particularly in terms of indicators and reference territorial scale, and perhaps a new medium-term objective defined.

5. The initiatives aiming to make the socio-economic issues of biodiversity known to different audiences are supported and developed.

# Chapter I

# General issues

## 1. Terms of reference

In a letter dated the 16th of January 2008 (attached to this report), following a proposal by the Minister of State for Ecology, Energy, Sustainable Development and Planning, the Prime Minister asked the Secretary of State to the Prime Minister with responsibility for Forward Planning and Assessment of public policies to:

1. *"Report on the fund of scientific knowledge on the subject of the monetarisation of services delivered by ecosystems, and the value of biodiversity,*
2. *Analyse the socio-economic issues of biological diversity in France, including the overseas departments and communities,*
3. *Propose terms of reference for any future research,*
4. *Estimate the first reference values for accounting for biodiversity, which may be used in particular for socio-economic studies relating to infrastructure projects".*

The reasons for this request were a reminder of the fact that the "*changes in* biodiversity were fundamental to the environmental concerns of our society", *and that,* in the context of the conclusions of the Grenelle de l'environnement [Roundtable on the Environment], *the* President of the Republic was commmitted, "to all future public decisions being made whilst taking the cost to biodiversity into account". In his closing speech at the Grenelle de l'environnement on the 25th of October 2007, the President of the Republic in fact also said: "Obviously, a project whose environmental costs are too high will be rejected (…) Ecological solutions will no longer have to prove their worth. Non-ecological projects will have to prove that it was impossible to do it any another way[1]"*.*

Viewed from this angle, the terms of reference put the emphasis on the need to "*be in possession of objective evaluation criteria that will enable us to take better account of the value of biodiversity and the services delivered by ecosystems. These evaluation criteria can contribute towards defining reference values for guiding the Government when making decisions*".

---

[1] Article 2 of the Government bill relating to the implementation of the Grenelle de l'environnement stipulates that procedures for making public decisions that might have a significant effect on the environment should be revised so that proof must be provided to show that it has been impossible to come to an alternative decision more favourable to the environment at a reasonable cost.

To respond to this request, the Centre for Strategic Analysis has set up a working group, presided over by Bernard Chevassus-au-Louis and made up of biodiversity experts, economists, union representatives, environmental associations and civil servants. Please see the list of members attached.

## 2. Emerging concerns

This desire to consider the socio-economic aspect of biodiversity and ecosystems is evidenced by developments in the international, European and national political context that were first seen around twenty years ago. It might be said that the symbolic starting point for such considerations was the Convention on Biological Diversity (CBD), signed in Rio in 1992.

### 2.1. The CBD and extensions thereof

Ratified by 168 countries (out of 191) on that day, the Convention on Biological Diversity was the first international convention on the subject of global biological diversity. The introduction states that: "the conservation and sustainable use of biological diversity are of the highest importance for meeting the food, health and other requirements of the planet's population".

However, the commitments made under this Convention are hardly restrictive, and involve a large number of approaches, which offers the signatory countries a wide margin for interpretation and manoeuvre. A clear example of this can be found in the wording of article 6, for example:

> *"Subject to its own situation and resources, each contracting party shall:*
> *a. develop national strategies, plans and programmes for the conservation and sustainable use of biodiversity, or adapt existing strategies, plans and programmes for this purpose, to include the measures announced by this Convention that might concern it;*
> *b. where possible and appropriate, build the conservation and sustainable use of biodiversity into the relevant industrial and pan-industrial plans, programmes and policies."*

It was ten years later at the Johannesburg Conference in 2002, that a more focused commitment – one based on a duty to deliver results, rather than simply to provide resources – was made by the signatory parties, namely: "between now and 2010, *achieve a significant slowing down in the current momentum of the impoverishment of biodiversity at a global, regional and national level* as a contribution towards reducing poverty for the benefit of all forms of life on Earth".

The French objective, which reflects that of the European Union, adopts the same period, but has greater ambitions. Established in 2004 as part of the "National Biodiversity Strategy", it proposes "stop the erosion of biodiversity between now and 2010".

**These various commitments are all affirmation of a strong link between biodiversity and development**: it is certainly the case of the CBD, which was drawn up as part of the more general Rio Convention on Sustainable Development; it also applies to France, since the national biodiversity strategy is part of the Stratégie nationale du développement durable [National Strategy for Sustainable Development] (SNDD), adopted in by the French Government in 2003. This is why is has seemed increasingly necessary to conduct a more detailed evaluation of the economic aspects of biodiversity and the services delivered by ecosystems.

## 2.2. European environmental directives

It is clear to us today that several past European directives relating to the environment already contained the seed of this concern for economic evaluation, even if only expressed indirectly.

From as early as 1985, **directive 85/337/CEE** relating to the evaluation of the effects of certain public and private projects on the environment and, later, **directive 2001/42 dated the 27th of June 2001** relating to the evaluation of the effects of certain environmental programmes, are based on the need to carry out evaluations, ultimately in economic terms, in order to quantify and reduce the impact on biodiversity and ecosystems.

Articles 6.3 and especially 6.4. of **Directive 92-43 dated the 21st of May 1992**, known as the Habitats Directive, relating to natural habitats and the conservation of natural habitats and wild flora and fauna across European territory, also cites the potential need for an ultimately economic evaluation of biodiversity. In fact, the following stipulations are made:

> *Article 6.3. Any plan or project not directly linked to or necessary to the management of the site, but which might affect such a site in a significant way, either individually or combined with other plans or projects, should be subject to the appropriate evaluation with regard to the effects on the site in terms of conservation objectives for the site (…).*
>
> *Article 6.4. If, in spite of negative conclusions being drawn from the evaluation of effects on the site, and in the absence of any alternative solutions, a plan or project still has to be carried out for reasons of major public interest, including social and economic, the member state shall take every compensatory measure necessary to ensure that the global coherence of Nature 2000 remains intact (…).*

A full and ultimately monetary evaluation of ecosystem services potentially affected by development might therefore prove necessary in order to ascertain an appropriate definition for “major public interest”.

In terms of aquatic environments, the **2000/60/CE outline directive on water** offers a globally integrated framework for managing continental waters with a view to reducing pollution levels, promoting sustainable use of water resources, conserving protected areas, improving the ecological condition of aquatic ecosystems and reducing the effects of floods and droughts. This integrated approach involves identifying all of the functions and weak points of aquatic ecosystems, which can in turn lead to the requirement for a monetary evaluation of the associated services, and the cost if they become impoverished. Furthermore, when member countries indicate that they cannot achieve the objective of restoring bodies of water back into good condition, they will have to argue from the standpoint of the disproportionate cost involved.

More recently, **directive 2004/35 dated the 21st of April 2004** on environmental responsibility relating to the prevention and repair of environmental damage also gives rise to further need for evaluation. There is also the **communication from the**

**European Commission dated the 22nd of June 2006:** "Arresting the loss of biodiversity in 2010 and beyond – Preserving ecosystem services for the well-being of Humanity". Accompanied by a road map promoting biodiversity in the next decade, it explicitly acknowledges that the motive force behind the loss of biodiversity is "the failure of governments and conventional economics to recognise the economic values of our natural assets and ecosystem services". Finally, we can quote the recent European Parliament resolution (2008/2210(INI) dated the 3rd of February 2009), which "asks the Commission to identify unspoilt areas of nature", requiring it to "conduct a study on the value and benefits of protecting unspoilt areas of nature" and to "devise a community strategy" ad hoc.

In the same vein, we can cite the thinking initiated several years ago by the European Union around the Eurovignette [Road Tolls]: updating a directive from 1999, directive 2006/38 dated the 9th of June 2006 gives member countries the option of imposing a tax or charge on trucks using the European road transport network. In accordance with the rules for standardisation laid down by the directive, member countries have the option of applying different tolls, depending on each type of vehicle, emissions category ("EURO" classification), level of damage that they cause to the roads and the location, time and level of disruption. At the current time, only three "negative external factors" have been identified in this way (atmospheric pollution, noise and congestion), and there is intense debate around the fact that other types of impact, especially on climate and biodiversity, have not been taken into account[1]. In France, motorway tolls are a response to this directive, and MEEDDAT calculations show that trucks cover the whole of their costs across the motorway network available to them in terms of the tolls they have to pay. However, these costs do not include any evaluation figure for the loss of biodiversity, as there is no evaluation method available. It should also be emphasized that, since 2001, Switzerland has not complied with this directive, but has applied a "Redevance sur les poids lourds liée aux prestations" [Service-related charge for trucks] (RPLP) since 2001 that takes these types of impact into account[2].

Finally, at national level, the obligation to provide each Government bill with an environmental impact study from September 2009[3] is still under discussion, and is just one more illustration of the move towards a priori evaluation of the impacts of public policies.

## 2.3. Methodology studies

To meet this diverse range of expectations, several methodology studies have been initiated by various bodies.

---

[1] A report on the methods for estimating different types of impact was recently produced at the request of the European Union: Mailbach M. *et al.* (2008), *Handbook on Estimation of External Costs in the Transport Sector*, CE Delft, Netherlands.

[2] The estimated external cost of road transport in 2005 is 9.3 billion Swiss francs, including 687 million for impact on nature and the landscape (source: Office fédéral du développement territorial ARE).

[3] Article 7 of the draft framework law concerning the application of articles 34-1, 39 and 44 of the Constitution after the first reading by the National Assembly and the Senate.

So, since 2001, as part of a mandate assigned to it by OECD environment ministers and in accordance with ruling IV/10 issued by the Conference of the Parties to the Convention on Biological Diversity, the **OCED working group on the economic aspects of biodiversity** focused its second guide to methodology on the issue of the monetarisation of biodiversity[1]. In particular, the guide listed nine possible political areas and objectives of evaluation, ultimately monetary, of biodiversity:

- To demonstrate the value of biodiversity - moral, aesthetic, economic and ecological - for greater social awareness,
- To establish the level of damage caused by the loss of biodiversity: aspects of environmental responsibility,
- To revise national economic accounting systems,
- To set levels for taxes and fines,
- To facilitate decision-making for land planning and use: promote sustainable agriculture and forestry, and set priorities for establishing protected areas,
- To demonstrate the cost of invasions biologiques with a view to better prevention,
- To limit or ban the trade in endangered species;
- To evaluate the impact on biodiversity of investment in road, airport and residential infrastructures,
- To set priority objectives for the conservation of biodiversity in the context of a limited budget.

This rather unstructured list clearly shows that requirements for the economic evaluation of biodiversity (and ecosystems) differ greatly from each other and involve decisions and action at a range of levels.

There was a major change in the thinking on methodology at the **Millennium Ecosystem Assessment (MEA)**, which ran between 2001 and 2005 and involved 1,360 experts from 95 countries. This study, which we will come back to at length, followed up work carried out by the Intergovernmental Panel on Climate Change (IPCC), which had identified the threats to our ecosystems resulting from climate change, and clearly was looking for a political angle to run alongside the climate change perspective. Published in 2005, the MEA report popularised the concept of ecosystems services, and suggested economic evaluations of these services for different ecosystems.

Since then, the work done by the MEA has been taken up again and incorporated in regional, national and local analyses. In 2008, MEEDAT initiated a similar approach for France in order to reach conclusions at a national level on state of ecosystems and the goods and services they provide and the value of such services and their contribution to human activities, whether they be productive, social or cultural. In the same vein, the European Environment Agency launched the **European Ecosystem Assessment (EURECA)** project in 2006, which covers the same logical framework as

[1] OECD, 2002, *Manuel d'évaluation de la biodiversité. Guide à l'intention des décideurs*, OECD Publications, Paris, 177pp.

the MEA, type of services supplied by ecosystems and the Streamlining European Biodiversity Indicators (SEBI), but goes into more detail, and is more directed at policy than the MEA. The results are due for publication in 2012.

Finally, **the TEEB process** (*The Economics of Ecosystems and Biodiversity*), initiated in March 2007 under the German chairmanship of the G8 Summit in Potsdam and coordinated by the economist Pavan Sukhdev, aims to calculate the global cost to society of the impoverishment of biodiversity and ecosystem services. Thus, it both continues the work of the MEA and adds a further viewpoint with the *"Stern Review of the Economics of Climate Change",* published in 2006, which carried out a similar study of the global cost of climate change, set against the cost of resisting such change. An interim report was published in 2008, and the final report is expected early in 2010.

A completely different area of politics creating a need to evaluate biodiversity and ecosystems is represented by the current review of the UN System of Integrated Environmental and Economic Accounting **(SEEA)**. One of the objectives is to include ecosystems and services as natural assets. In France, the **Commission des comptes et de l'économie de l'environnement**
**[Commission on environmental accounting and economy] (CCEE)**[1] has been set up, following the model of other industrial commissions (transport, tourism, agriculture, etc.). Here, the main aim is to inspect and approve the *Annual Report on the accounts for the environmental economy* produced by the French Institute for the Environment (IFEN)[2]. These accounts for the environmental economy currently represent a "satellite" area in terms of the national income accounting system, and are essentially based on reporting the costs of protecting and enhancing the environment. In due course, environmental expenditure accounts, which are at the core of the accounts for the environmental economy, will become an integral part of the SEEA.

We might connect these particular concerns with the "Commission on the measurement of economic performance and social progress", set up by the President of the Republic in 2008, and chaired by Joseph Stiglitz, Nobel Prize winner for economics. This Commission has a mandate to identify the limits of the GDP as a progress indicator in terms of sustainable development, taking into account the negative impact on the environment.

## 3. The two levels of approach and the guidelines of the report

In terms of the work being carried out, it appears to be possible, but above all necessary, to make a distinction between two major areas of concern and public activity:

- **A "macro-economic goal"**: efforts are being made to find an approximate value for biodiversity and economic services at the level of a single country, large bio-geographical groups and even the entire planet, and set global objectives and develop instruments for management and governance at this level. These instruments often require strong involvement from the public

[1] Missions defined in article D.133-35 of the French Environmental Code.
[2] In November 2008, became the SOeS (Service de l'observation et des statistiques) for MEEDAT.

authorities: major international conventions (on sharing the benefits linked to the use of biodiversity, for example), and extensive ecological infrastructures.

This is the same type of approach as the one that, in terms of greenhouse gases, addresses the issue of the global consequences of climate change. This was the central objective of the Stern report, and can also be found in the TEEB process. The issues of environmental compatibility and critical GDP analysis are also linked to this particular concern.

- A **"micro-economic goal": how do we include the impacts on biodiversity in the costs for each project or activity**, so that the socio-economic evaluation of its viability takes account of this factor on a consistent basis at national level? It's strongly related to the Eurovignette initiative and the "Boiteux II" report by the Commissariat général du Plan [Economic Planning Commission] (2001), which proposed reference values for inclusion in noise and pollution evaluations, as for time gained or carbon emissions or savings. The report failed to mention the issue of biodiversity (which was not even mentioned as an issue that would be addressed in further study …), which was tantamount to giving it zero value in the socio-economic calculations used to optimise decisions, particularly about infrastructures.

**The Prime Minister's referral is clearly linked to this second concern**, even if the "public decisions" referred to do not solely address infrastructure projects. **It is therefore on this particular micro-economic area that the working group has focused its analysis, whilst still considering the fact that all of these micro-economic decisions would need to be consistent with global objectives for conservation of biodiversity, with particular emphasis on halting the erosion of biodiversity between now and 2010.** In particular, this objective requires that the economic evaluation of biodiversity should follow a logical system of "strong sustainability", **meaning that it should not be used for transactions with other elements of well-being that might compensate for losses of biodiversity.**

It would also seem useful to make a distinction between ex-ante concerns, when a project – or a range of project options – is examined in terms of the effects on biodiversity, and ex post concerns, when faced with wilful or accidental damage that we are trying to evaluate and, ultimately, ensure that those who have caused it remedy the situation.

The two issues are noticeably different, especially as, in the second case, the estimate of "reparations", namely, legal rulings, can mainly include the concepts of "social disapproval" and "moral prejudice" as bases for evaluation. As we will see, this does not mean that such considerations (and other non-economic considerations) are inadmissible in terms of *analyses carried out* ex ante.

**The essential focus of this report is *ex-ante***, i.e. focused on defining reference values that will enable biodiversity to be included in the public decision-making process. But we also thought it useful to look into ex post approaches. This is why we have attempted to establish how the law addresses these issues, especially the methods used to attribute an economic value to losses associated with the environment. It is with this in mind that we will also be looking at compensation practices currently adopted abroad, and which are beginning to appear in France.

## 4. The major stages of the report

Please find the full text of the terms of reference and the list of group members attached.

Against the background of the four issues raised, the following stages have been adopted:

- First of all, it seemed necessary (chapter II) to give a summary of the major socio-economic issues represented by biodiversity and ecosystems services, not only today, but in the future, with an emphasis on France's serious responsibilities in this area. Chapter III then demonstrates how the Law has progressively addressed these issues, and how it has handled them.
- A major development was considered necessary in order to describe the scientific concepts included in the notions of biodiversity, ecosystems services and the links between the two (chapter IV). This chapter also outlines the various indicators that can be used to identify the current state of biodiversity, its development and the pressures weighing upon it. It ends by detailing the hypotheses reached by the group, which can be used to create a dialogue between biological concepts and economic analysis.
- Chapter V reports on levels of scientific knowledge in terms of evaluating ecosystems and biodiversity. Specialists in these disciplines will not need to read the first section, but, given the serious debate around the legitimacy of the economic approach itself in this field, the working group thought it necessary to go back to the bases of the economic theory of value. This chapter then goes on to describe the extensions of the framework for analysis that will be required to address biodiversity, then the different methods that have been developed, in order to arrive at an approximate value for goods that are not the subject of direct market transactions, finally giving the principle published results relating to the component factors in the value of biodiversity. It concludes by mentioning the contributions made by several initiatives recently led by institutions in relation to the subject of this report.
- Following these two chapters, which describe the current “state of play”, chapter VI gives the main important areas for research identified by the group as they carried out their studies.
- Based on these theoretical approaches, chapter VII addresses in detail the technical aspects of establishing reference values, pointing out that there are currently major differences between the issue of biodiversity and that of carbon sequestration, which is often cited as a reference. It offers several concrete illustrations, particularly for temperate forests, and examines how the use of such values can influence changes of land use. It clarifies procedural issues that arise in both identifying and using reference values, so that this approach can be taken on board by all the stakeholders. Finally, this chapter describes the non-monetary approaches that might be used, with particular reference to compensation practices;

- General conclusions assess the principal responses to the four issues of the referral, and give a number of directions and recommendations for using and continuing the work.

It should be said right from the start that, in line with its mission statement, the group essentially addressed the issue of establishing reference values. Nevertheless, it has limited itself to setting up a number of reference points – which seemed to be particularly important – with regard to the scope of and possible methods for using these values. However, it believes that the way in which public action - via a range of instruments (regulations, taxes, incentives, etc.) - integrates these approaches and combines them with other concerns, is crucial, and must therefore be explored and clarified.

In fact, the group takes the unanimous view that it would be unacceptable for the approaches that it has agreed to examine to be, paradoxically, applied in order to take less account of the issues of conservation and the sustainable use of biodiversity and ecosystems services, the importance of which is now widely recognised.

# Chapter II

# Biodiversity: the socio-economic and political issues

## 1. Why is biodiversity so important to society?

### 1.1. Biodiversity – the fruit of interrelationships between living things

Definitions for the neologism “biodiversity” are many and varied and the scope of its significance for society immense. In fact, we would do well to consider the *totality of living things in interaction, including micro-organisms and the services delivered by ecosystems”* (Babin *et al.*, 2008). The level of biodiversity we see today is the result of billions of years of evolution, created by natural forces and suffering the influence of Man at ever increasing and accelerating levels over the last five decades. **Biodiversity, and the ecosystems at the heart of which it is found, deliver a great number of services that support human life:** food, fuels and building materials; the purification of air and water; the stabilisation and control of the world’s climate; the control of flood, drought, extreme temperatures and the force of the wind; the generation and renewal of soil fertility; the maintenance of genetic resources that contribute to plant variety and animal species, medicines and other products; plus cultural, recreational and aesthetic benefits.

At a global level, **biodiversity should be considered “*in terms of its relationship to major issues, such as the reduction of poverty, food and drinking water for all, economic growth, conflicts associated with the use and appropriation of resources, human, animal and plant health, energy and climate change*.** *This vision requires that we link biodiversity to human well-being as we strive to reach development targets for the millennium”* (Babin *et al.*, 2008; UNDP, 2004). In a world that is undergoing rapid change in response to the effects of accelerated changes in land use and climate conditions and the upheaval of human societies in terms of their relationship with the environment, biodiversity is now recognised as “life insurance for life itself” (McNeil and Shei, 2002). The United Nations made it the theme of their International Biodiversity Day on the 22nd of May 2005, under the heading of ”**Biodiversity – life insurance for a changing world**”.

Thus, biodiversity is now no longer seen only through the lens of nature conservation for its own sake, or of saving certain symbolic species - far from it. Human societies, even the most advanced ones, are now aware of the interaction between human beings and the biodiversity of which they are part

***Endogenous dynamic interaction* – the watch-words of life.** We need to interact to cooperate, procreate, change the environment in which we evolve and adapt to its own natural developments. Likewise, it is vital that we interact with the whole of the living world: we only consume living organisms - vegetables, fruit and meat – and cooperate with living organisms to obtain products from the fermentation process: beer, wine, cheese and sausage, amongst others. Our habitat is largely made up of materials from the living world. We have also inherited fossil and limestone deposits from biodiversity long since gone, as we have the Earth's atmosphere. Our health depends on biodiversity. It is estimated that three quarters of the world's population depend on traditional natural remedies. In China, of the 30,000 superior species of plants recorded, over 5,000 are used for therapeutic purposes. Nearly half[2] of all synthetic medicines come from natural sources and, of the anti-cancer medicines, 42% are natural in origin (Newman and Cragg, 2007).

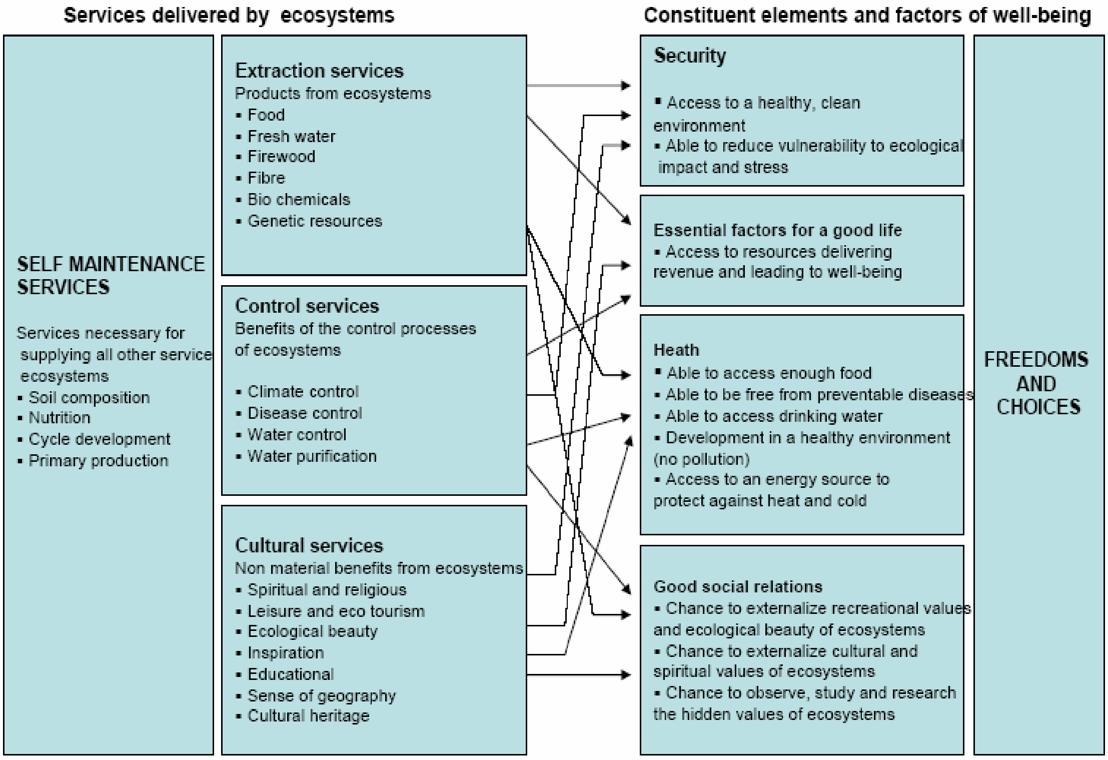
**The main difficulty in understanding the issues in terms of socio-economics and biodiversity is firstly how to identify the full effects of the latter on human daily life: it is everywhere, from food for digestion, to protecting skin, to the chemical industry**. An added difficulty is the dynamic nature of biodiversity, which requires that we take the time-duration parameter into account.

### *1.2. Biodiversity as support for ecosystem services*

The Millennium Ecosystem Assessment, 2005a, published in 2005, has had a considerable impact in terms of its proposal for a shared framework for thinking about the issue of ecosystems in connection with social well-being, its definition of the concept of "ecosystem service", also known as "ecological service" (humans make free use of the properties of such ecosystems) and the creation of a system for typing these ecological services.

**Ecosystem services derive from interactions between the organisms that shape environments and their function within the ecosystems. The purification of air and water, carbon sequestration and soil fertility are services that result from interaction, rather than organisms themselves.** Each ecosystem (forest, wetland, grassland, coral etc.) has different functions and services, which in turn depend on the health of the ecosystem, the pressures weighing upon it and the way they are used by society in a given bio-geographical and geo-economical context (see sections II.4. and II.6.3). Human society uses ecosystems and, through this, they change them at local and global level. In turn, society adjusts its habits to perceived change. This dynamic interaction is what it is appropriate to call **socio-ecosystems** (Walker *et al.,* 2002).

**Figure II-1 : The benefits derived from ecosystems and their links with the Human well-being**




*Source:* Millennium Ecosystem Assessment, *2005*

**Likewise, the perception and use of ecosystem services largely depends on the scale considered.** For example, the benefits derived from the non-ligneous products of a forest, such as berries and mushrooms, is of a rather local or regional interest – even where some are gathered by a big organisation employing unskilled labour– whilst the importance of the forest as a source of carbon is of global importance. A study conducted at the request of the Department for Environment, Food and Rural Affairs (EFTEC, 2005) on the economic, social and ecological values of ecological services focused on this concept of scale in terms of how services delivered by ecosystems are perceived. Based on the bibliographical summary of a number of case studies compiled at international level, the study classifies services in relation to three major types of ecosystem (forests, wetlands, agro-ecosystems), in terms of their use values in relation to three levels of benefit: local, regional/national and global (see tables II-1, II-2 and II-3). **Thus, the intricate relationship between the scales associated with ecosystems services makes it difficult to attribute monetary values to ecosystem goods and services.**

Table II-1: Total economic value of forest goods and services

| | Goods and services | Local | Regional | Global |
|---|---|---|---|---|
| Direct use | Forest products<br>Wood<br>Charcoal<br>Non-ligneous products | <br>X<br>X<br>X | <br>X | <br>X |
| | Genetic resources<br>Traditional medicines<br>Pharmacology<br>Research | <br>X<br>X<br>X | <br>X<br>X<br>X | <br>X<br>X<br>X |
| | Recreation and tourism | X | X | X |
| Indirect use | Control of local rainfall | | X | |
| | Control of flooding and water supply | X | X | |
| | Control of soil erosion | X | X | |
| | Carbon sequestration | | | X |
| | Health | X | | |
| Options | Future direct/indirect use of goods and services mentioned above | X | X | X |
| Non-use | Traditional knowledge and culture | X | X | X |

*Source: EFTEC-DEFRA 2005 (adapted)*

Table II-2: Total economic value of goods and services from wetlands

| | Goods and services | Local | Regional | Global |
|---|---|---|---|---|
| Direct use | Stock rearing/crops | X | X | |
| | Fishing | X | X | |
| | Fibre for construction, craft production and heating wood | X | | |
| | Hunting water fowl and other wild animals | X | X | |
| | Aesthetic value of wetlands, recreation | X | X | X |
| Indirect use | Control of storms | | X | |
| | Retention of high water levels and regulation of flow | X | X | |
| | Recycling of sediments and nutrients – improvement of water quality | X | X | |
| | Control of erosion by vegetation | X | X | |
| | Carbon sequestration – reducing climate change | | | X |
| Options | Future direct/indirect use of goods and services mentioned above | X | X | X |
| Non-use | Life/heritage/altruistic values of habitats and species linked to wetlands. Traditional knowledge and culture; traditions | X | X | X |

*Source: EFTEC-DEFRA 2005 (adapted)*

Table II-3: Total economic value of goods and services from agro-systems

| | Goods and services | Local | Regional | Global |
|---|---|---|---|---|
| Direct use | Crops/food | X | X | X |
| | Stock rearing/food | X | X | X |
| | Agricultural landscape amenities | X | X | |
| Indirect use | Control of pests and diseases | X | X | |
| | Soil-related processes<br>- mineralisation process<br>- preservation of soil structure and porosity<br>- preservation of soil fertility | <br>X<br>X<br>X | <br>X<br>X<br>X | |
| | Pollination | X | X | |
| | Cycle of nutrients | X | | |
| | Carbon sequestration | | | X |
| | Water quantity and quality | X | | |
| | Genetic diversity | X | X | X |
| Options | Future direct/indirect use of goods and services mentioned above | X | X | X |
| Non-use | Traditional knowledge and culture; traditions | X | X | X |

*Source: EFTEC-DEFRA 2005 (adapted)*

A number of studies have appeared in support of ecosystem services, at the same time demonstrating the cost to society of losing such levels of biodiversity.

### *a. The cost of the collapse of marine fisheries*

Work carried out by the Centre for Fisheries Research at the University of British Columbia demonstrates the impact of the collapse of fisheries on a global scale from an economic, employment and health point of view (Pauly *et al.,* 1998, 2003). Free access and perverse subsidies are the key things that explain this collapse, with over-exploitation of over a quarter of global marine fisheries and maximum thresholds reached for half of them. The risks are considerable in economic terms, as this activity represents a revenue level of around 100 billion US dollars a year and 27 million jobs, but above all in terms of public health, as, for over one billion human beings, fish is their only source of animal protein, especially in developing countries.

### *b. The importance of biodiversity for pollination*

A recent study published in the *Ecological Economics* Review (Gallai N. *et al.*, 2009), based on an extensive bibliographical review of the dependence of major food crops on pollinators and FAO data from 2007, has set the value of the service delivered by pollinators at 150 billion Euros. The main crops in question are fruit (50 billion), vegetables (50 billion) and oleaginous crops (39 billion). The average value of crops dependent on pollinators (760 Euros per tonne) is much higher than that of non-dependent crops such as cereals and sugar cane (150 Euros per tonne). If the pollinators were to disappear, the global balance between foodstuffs would be radically changed in terms of fruit, vegetables and stimulants (coffee and cocoa). This would particularly affect importing regions such as the European Union.

In addition to domestic pollinators such as the domestic bee *Apis mellifera,* wild pollinators such as wild bees, butterflies, diurnal butterflies and moths, wasps and also certain birds and bats play a crucial role in agricultural production (Klein *et al.*, 2007; Balmford *et al.*, 2008). Furthermore, these pollinators are important for wild flora, which in turn plays a part in maintaining other ecosystem services.

### *c. Health risks linked to the disturbance of ecosystems*

The loss of biodiversity makes great changes to the balance of interactions within communities of species, especially to the predator-prey relationship and host-parasite and symbiotic relationships, but also to interaction between pathogenic agents, vectors and hosts. Pathogenic agents of infectious diseases are an integral part of ecosystems, within complex networks with other organisms that govern their emergence, transmission and propagation. Noting that in 60% of human infectious diseases, pathogenic agents have developed and been living on other organisms before reaching humans, Molyneux *et al.* (2008) are able to demonstrate the direct consequences of the loss of biodiversity linked to the impoverishment of the ecosystems in the propagation of these diseases.

Convergent experiments carried out in Brazil and the United States prove that biodiversity is an important factor in the inhibition of numerous diseases (Leishman's disease, Chagas disease, Lyme's disease, etc.). On the other hand, the destruction of these environments is a factor that promotes the propagation of these diseases (Laffitte and Saunier, 2007).

## 1.3. Biodiversity as support for sustainable agriculture

The intensive agriculture and rearing methods that developed over the second half of the 20th century in response to the increase in the global demand for food could, it seems, be considered as hardly dependent on biodiversity: they are characterised by a weak species (monoculture), genetic (uniform varieties) and ecological (homogenised environment providing economies of scale) diversity. Apart from the fact that this an erroneous assumption – you only need to remember that they often use an excess of products from historic biodiversity, whether it be oil and natural gas, phosphates or the wild genetic resources required for the creation of modern varieties – **it would seem today that these intensive systems are likely to come up against a good number of limits to sustainability in the course of the 21st century.**

These limits are particularly in terms of their environmental impact, the inevitable rise in input costs (water, fertilizer, and energy) and their acceptability by society. Furthermore, expectations of the role of agriculture have changed significantly: if the aim of increasing global food production (double between now and 2050) remains full of possibility, the reduction in the agricultural population associated with mechanisation, which saw the industry and related services soar in developed countries throughout the 20th century, hardly seems desirable, and is even to be avoided in developing countries. On the other hand, besides controlling these impacts, **a positive contribution from agriculture to the production of the ecological services that we have just mentioned is now greatly desirable**, given the predominance of agricultural activity (including stock rearing and forestry) in land management.

The issue of developing new types of agriculture that are productive in the economic, social and environmental sense, is therefore a major one for the years to come. **It seems that these new types of agriculture will have to redefine themselves on the basis of a detailed understanding and appropriate use of biodiversity:** the design and creation of more diversified plant and animal populations (mixtures of varieties or species) to make better use of local resources and resist pests and diseases; the reorganisation of the land in order provide conservation of water resources, fight against erosion, maintain useful biodiversity and manage soil biology to mobilize fertilizing components better are just a few examples of the new approaches (see Griffon, 2006, or Chevassus-au-Louis and Griffon, 2007), which should encourage more and more research, training and agronomic development at a global level.

The recent collective report on the subject of "Agriculture and biodiversity: valuing the synergies" (Le Roux *et al.*, 2008), based on an analysis of over 2,000 bibliographical references, shows how **biodiversity is the basis for three levels of services to agriculture:**

- Services contributing directly to agricultural incomes such as yields and product quality. Thus, on grassland, benefits can be obtained from complementary functions between species, notably with leguminous plants, which fix nitrogen from the air, or crop rotation. The botanical diversity of grassland and ranges has a definite effect on stimulating the appetite (and therefore the digestion) of domestic herbivores. Furthermore, the presence of certain grassland species can improve the organoleptic quality of cheeses.
- Services contributing the smooth functioning of agro-ecosystems via:
    - biological control (role of the natural enemies of pests, pollinators…). Bees and syriphid flies are key insect groups for these types of services. The preservation of "source" populations requires favourable land conditions, combining diversity of coppices and type of land area and the presence of semi-natural habitats;
    - the supply of resources to plants (fertility, physical stability of the soil), i.e. the conditions in terms of water supply and mineral elements for crops, ensured by the biodiversity of the fauna and the micro-organisms in the soil, but also that of the flora;
- Services not directly related to agricultural revenues: limitation of pollution of water tables, climate regulation (carbon sequestration by grassland, for example), shaping the landscape, rural tourism supports.

The inclusion of the concept of agro-ecosystem services, together with that of the multi-functionality of agro-ecosystems, is at the heart of the thinking behind payments to farmers in return for ecosystem services, as promoted by the FAO (FAO, 2007).

## 1.4. Biodiversity as support for a new "bio-technology"

The report to Parliament and the Senate by senators P. Laffitte and C. Saunier (2007) hammers it home: "**The sustainable valuation of biodiversity is a necessity, but also a chance we must grab with both hands. Biodiversity can be one way of supporting changes to our economic development methods, changes which have**

**been made unavoidable by the energy and climate crises that we are now experiencing”.** Their report places particular emphasis on:

- The services delivered by ecosystems.
- The reserve of assets we have – the box of tools that can be used in a fourth industrial revolution based on bio-mimetics, bio-inspiration, valuing bacterial biodiversity and bioprospection.

**Bio-mimetics consists of** recording remarkable behaviour, understanding the relationship between behaviour and structure, and imitating this structure to develop sustainable materials cheaply and quickly. Examples of application in industry are (Benyus, 2002):

- The development of a self-cleaning exterior paint inspired by the astonishing properties of the surface of the lotus leaf (dirt does not adhere). Rain water is enough for the painted surface to maintain its initial brightness, and chemical cleaners are no longer required.
- Anti-friction surfaces adapted to modern electrical systems, inspired by the amazingly slippery skin of the sand fish, a lizard from the Arab Peninsula.
- An avant-garde system for collecting water, enabling buildings to capture the water they require from damp air. This system was inspired by the way the scarab beetle in the Namibian desert captures water from coastal fog.
- Biodegradable packaging sleeve that waterproofs pipe systems by imitating the way a frog in the Australian desert keeps a film of water around it while waiting for the rainy season.
- The possibilities are endless, as shown by A. Guillot and J. A. Meyer (2008). In partnership with the UNEP (United Nations Environment Programme) and the IUCN (International Union for Conservation of Nature), the Biomimicry Guild and the ZERI (Zero Emissions Research and Initiatives) research group have drawn up an initial list of some 2,100 examples, which they will be using to prepare a work entitled *Les 100 meilleures astuces et leçons de la nature*.[*The 100 Best Hints and Tips from Nature*]

**Bio-inspiration**, which is at a less advanced industrial stage, will attempt to identify molecules with certain properties and derive various objects from them that are different from those created in the living world.

**The valuing of bacterial biodiversity** opens up new horizons as a substitute or support for chemistry: either for direct production (example: penicillin), or by extracting enzymes to facilitate bioconversions to make them complete one stage of a chemical process. There are numerous examples of industrial use: the manufacture of Tergal, pollution control, beta-carotene manufacture, production of methane and corticosteriods etc.

**Bio-remediation**, a group of new techniques using micro-organisms and plants to decontaminate soil or water, has proved promising in economic terms. It is currently responsible for turnover of over 100 million Euros, and could see a global increase to 10 billion Euros between in a few years. In France, it could help to clean up 20,000 sites (CSPNB, 2007).

## 2. What is the status of biodiversity? What are the risks and threats that it faces?

We cannot analyse the socio-economic and political issues relating to biodiversity without carrying out an up to date evaluation of the exceptional heritage of France and it rich biodiversity. But we must also carry out a proper assessment of the losses already incurred and the risks involved in continuing in certain directions, and of new threats.

### 2.1. France: a country unaware of its amazing diversity

Levels of biodiversity in Metropolitan France might seem remarkable, but **the biodiversity of its overseas territories puts France in a unique position in the world, and this brings with it a huge responsibility: together with its overseas communities, France holds over a third of the world's recorded species.** France is the only country with a presence in 5 of the 25 biodiversity hot-spots (Mediterranean, Caribbean, Indian Ocean, New Caledonia and Polynesia) and in one of the three major forest zones on the planet (Amazon). At 11 million km$^2$, it covers the 2$^{nd}$ largest maritime area in the world. France has 10% of the coral reefs and lagoons on the planet, with 14,280 km$^2$ of coral reefs, lying 4$^{th}$ behind Australia, Indonesia and the Philippines. The originality of the flora and fauna of New Caledonia, which is no bigger than Picardy (three departments), is comparable to that of the whole of continental Europe; on a surface area equivalent to just a few Paris arrondissements (40 km$^2$), Rapa island in French Polynesia is home to at least 300 endemic species (Gargominy, 2003) – i.e. naturally confined to a given region.

**Metropolitan France is a biological crossroads for Europe, with four of the eight principal bio-geographical zones** (Atlantic, Continental, Mediterranean and Alpine). By this calculation, it shelters 57% of the habitat types of Community interest listed in the Habitats Directive (CTE/DB, 2008) and 40% of the flora of Europe on less than 12% of its surface, with a major proportion of endemic species, especially in its Mediterranean and alpine regions (IFEN 2006, IUCN France, 2005). The rural areas, where agricultural and pastoral activities in particular are carried out, are still more developed, covering over half of France's territory. It is here that not only all of the wild habitats and species of the territory can be found, but also the unique living heritage of cultivated or bred species - varieties and breeds selected throughout history. This rich heritage comes in the form of the identification, across the entire metropolitan territory, of nearly 15,000 ZNIEFFs [natural areas of ecological interest for flora and fauna], including 1,921 type II ZNIEFFs (large natural areas rich in diversity and hardly changed, offering biological potential) and 12,921 type I ZNIEFFs (areas of great biological or ecological interest) (INPN-MNHN, 2008).

**The extreme variety of its mainland landscapes greatly contributes towards making France the number one tourist destination in the world.** As underlined by the European Convention on the Landscape, ratified by France on the 17$^{th}$ of March 2006, "*the landscape is important in forming local cultures, and represents a basic element of the natural and cultural heritage of Europe, contributing towards the flourishing of human beings and the consolidation of European identity"* More than on any other continent, the biodiversity of Europe is the result of the interaction over thousands of years between human society and nature, continued in relative stability up until the industrial revolution, in spite of periods of intensification, alternating with periods of abandonment of the land (European Environment Agency, 2006). **Today, the rural landscapes that are home to the highest levels of biodiversity are firstly areas that are not favoured for cultivation (wetlands, very stony ground, ground that is too steep) and secondly agricultural areas that are worked extensively but not intensively, within small to average sized parcels of land, forming a patchwork of habitats working in conjunction with each other (areas of enclosure, traditional orchards)** (Collectif Science et Décision, 2007). Due to the intensification of agricultural practices and, inversely, the abandonment of agricultural activity in fairly mountainous regions, combined with increasingly diffuse urbanisation and the expansion of infrastructures, **the general trend is one of the homogenisation of rural landscapes, which has a direct impact on biodiversity** (Burel and Baudry, 2003, Lepart, 2005). **Specialist species are declining in favour of generalist species** (see section III.3.3).

In his study of the feasibility of “MEA France” (Levrel, 2007), H. Levrel emphasises the benefit of a landscape-based approach as a way forward for considering the integrated evaluation of ecosystem services and links with biodiversity management, and this is confirmed by a range of work carried out at European level (Mander *et al.,* 2007; Pedroli *et al.,* 2007; Marangon and Visantin, 2007).

## 2.2. The accelerated erosion of biodiversity…

At a global, over the last two hundred years, the rate at which species are disappearing is, according to the species in question, 10 to 100 times greater than the natural rate of extinction (1 in 50,000 species per century). By 2050, the rate at which particular species disappear could rise to between 100 and 1,000 times higher than the natural rate (Millennium Ecosystem Assessment, 2005b). **There is a growing consensus in the scientific community that we can even talk of a process moving towards a sixth extinction of biodiversity, almost solely due to the effects of human activity,** referring to five previous extinctions that have marked the living world through natural processes over a period of four hundred and forty million years back to the date of the first known extinction.

**France's heritage of living species is vulnerable because France ranks 4th in the world for endangered animal species and 9th for plants, according to the IUCN world danger list.** Of the 135 mammalian species breeding on French territory (including marine mammals), 49 are seen as endangered to various degrees. Of the 276 species of birds nesting in France, 51 are considered to be endangered. A number of examples illustrate the level of impoverishment: the area covered by French grassland has decreased by 30% in thirty years, 60% of wetlands disappeared in the 20th century, 75% of rivers contain pesticides, 50% of land is polluted by nitrates; in thirty years, the Beauce region has lost over 30% of the organic elements from its soil, and urbanisation is reported to have destroyed 800m$^2$ of natural spaces on French territory by 1980 and 60% of coasts have been urbanised, compared to 39% in 1960, and the arid and semi-arid coastal forests on La Réunion and New Caledonia have all but disappeared (Gargominy, 2003; UICN France, 2005; IFEN, 2006).

**The first evaluation of the conservation status of species and habitats of interest to the Community carried out in application of article 17 of the Habitats Directive**

**shows that 76% of the habitats in question in France are in an unfavourable state of conservation, and 41% in a bad state of conservation.** The situation is a little better in terms of species, with 50% in an unfavourable state of conservation and 31% in a bad state. The uncertainties around the state of 30% of species still leads us to believe that the situation could be worse than the evaluation actually shows (CTE/DB, 2008b, European Commission - DG Environnement, under pressure).

The insular nature of most of the overseas communities is demonstrated by an extremely high level of endemism, but also represents a fragility factor. Limited areas of population make them more vulnerable to extinction. **Thus, 30% of extinctions of mollusc species occur in overseas communities, putting France second behind the United States in terms of recognised extinctions.** The Mascareignes archipelago (La Réunion, Mauritius and Rodrigues) is often cited as an example of serious changes brought to island ecosystems by humans (Gargominy, 2003).

## 2.3. ...affected by growing pressure

In France as for the rest of the world, **five major types of pressure on biodiversity are seen to be the cause of its decline** (UNEP, 2006, *Collectif d'experts pour un état des lieux de la biodiversité* 2003):

- **The destruction and impoverishment of habitats**, caused by agriculture (agricultural intensification, abandonment of the land, drainage, irrigation), forestry (intensive use, replanting with single species), industrial fisheries and aquaculture, the construction of infrastructures and urbanisation leading to habitat fragmentation, developments for tourism, industry and mining …
- **Pollution** (eutrophication of aquatic environments, deposits of atmospheric nitrogen dioxide on vegetation, acidification of the soil, pollution by pesticides and heavy metals).
- **Invasive species**, where risks are increasing due to a strong increase in introductions, either deliberate or accidental, of species into environments different to their place of origin, particularly due to an increase in the number of routes of communication (roads, railways, planes, sea) and the intensification of trade.
- **Climate change.** It is estimated that a 1°C increase in temperature moves the line of tolerance for terrestrial species an average of 125km towards the poles and 150m towards the mountains. This is likely to bring major changes to the structure and functioning of ecosystems, and endanger those species which are not capable of extending their area of distribution quickly enough. The warming of marine waters is also a danger to a number of species: the increase in temperature already recorded is likely to play a part in the extensive bleaching of coral observed in Polynesia and the Caribbean.
- **The over-exploitation of wild biological resources** (hunting, fishing, gathering, exploitation of woodland) at a rate that is incompatible with renewal. Pressure by hunting is yet another factor in the decline of certain vulnerable species (especially birds). The problem is more acute in certain overseas communities (poaching of marine turtles in the Antilles, Mayotte, La Réunion and French Polynesia, in particular), where there is also the

problem of the over-exploitation of the trade in wild animals (traffic between Guyana and the Antilles, for example). The general state of a good number of stocks fished for in the Atlantic by French fleets is of concern (red snapper, skate, cod, pollock, monk fish, sole, langoustine, herring, mackerel etc.). Overseas, a number of different marine resources are being overexploited, especially in the Antilles (spiny lobster, white urchin, spider conch etc.), Mayotte and La Réunion (reef species, shells, spiny lobsters, sharks) (Gargominy, 2003).

The various pressures on biodiversity mainly result from the way our society uses space and natural resources in its choice of land development, production and processing options. **The underlying causes of the decline of biodiversity are therefore mainly socio-economic in nature.**

## 2.4. Gaps in knowledge and uncertain responses in the face of risks and threats

**However, there are still gaps in our knowledge of biodiversity, its relationship with the functions and services that human societies procure from it, its true extent and the processes governing its development, all of which serves to complicate any evaluations to support decision-making.**

If we consider only the "species" aspect of biodiversity, of the 10 to 15 million estimated species, only 1.8 million (12% to 18%) have been identified and named, and around 16,000 new species are identified every year. Knowledge in terms of taxonomic groups is extremely diverse: for vertebrates (fish, amphibians, reptiles, birds and mammals), which represent – with around 50,000 species - only a tiny proportion of biodiversity, it is estimated that 95% of existing species have already been identified, whereas only 10% of insects are actually known, essentially in the temperate zones. Most of the gaps in our knowledge relate to tropical forest invertebrates, marine organisms that live at great depths and micro-organisms. So, **the "unknown" area of biodiversity is certainly the biggest, not only in terms of species numbers, but also in terms of biomass and evolutionary divergence, and therefore function** (for example, on the African savannah lands, the total biomass of termites is greater than that of the large herbivores). Many people see this as the "hidden base of the iceberg", which runs the biosphere through its role in the cycles of organic and mineral matter, the control of populations and even as a life support system for the "big animal and plant species" via microbic symbiotic processes.

Political decisions on the protection of biodiversity are largely related to the concept of historic legacy, whether they concern protected species at national level or species and habitats of interest to the Community; and this is due to their legal status. But, **even for this limited number of species and habitats that are held as a legacy, it is noticeable that there are gaps in the data available in France in terms of national distribution, numbers, population dynamics and state of conservation.** This is particularly striking on reading the summary report by the European Commission (being finalised) relating to the conservation status of around 1,800 types of habitat and species of interest to the Community, as reported by the 25 member countries of the European Union under article 17 of the Habitats Directive.

In addition to our knowledge of species and their evolution, a number of scientific questions remain as to the place and role of individual species and groups of species in terms of the function, performance and resilience capabilities of ecosystems. How far do species complement each other to ensure that the ecosystem works? Are some of them not required in terms of delivering useful ecological services to society? If they are in fact redundant, are they actually a type of "life insurance" in case other species disappear or decline, where they fulfil similar functions? In more general terms, **we have only a limited understanding of the dynamics of ecosystems and resilience thresholds of ecosystems in the face of repeated stress.** The future of biodiversity subject to interference might therefore suffer extreme scenarios, from the most optimistic ones citing the resilience of ecosystems and functional redundancy between species, to the most pessimistic ones based on the domino theory.

**These questions become particularly acute when viewed from the perspective of the massive upheavals linked to climate change, thus raising a major socio-economic issue. In particular, this raises the issue of the ability of ecosystems to adapt to rapid environmental change and thereby lessen the effects.**

For example:

- The trend is for harvests to be earlier and earlier: grape harvests are one month earlier than at the end of the Second World War (García de Cortázar Atauri *et al.,* 2007).
- There is a tendency for species in interaction with each other not to be synchronized any more. Thus, the caterpillars that blue-tits feed on emerge earlier with the earlier springs. Not finding anything to feed on because the trees have not yet formed enough leaves, they have declined in number, which is detrimental to the blue-tit population, which declines in turn (Blondel, 2004). However, some species do manage to re-synchronize with each other. Orchids have blossomed in the Mediterranean a month earlier than usual, but also a month longer, waiting for the pollinators to arrive before wilting (Feldmann, 2007).
- In forest ecosystems, a team led by Christian Körner (Körner, 2005) has shown that the increase of $CO_2$ in the atmosphere created threshold effects beyond which trees would have a shorter life cycle.
- Warmer winters and longer growing periods can cause an explosion of certain "pests", which are even more damaging because the trees have already been weakened by drought (Rouault *et al.*, 2006).
- In the oceans, with fishing mainly focused on carnivorous species – species at the top of the food chain - ecosystems are being greatly affected, and more and more species with a shorter life span are being observed, such as squid and octopus (Cury and Miserey, 2008). In the North Sea, warm water species have already been observed, and some of them are now in competition with local species, such as mussels (Marbaix and van Ypersele, 2005).

Such changes to exploited ecosystems weigh heavily on for socio-economic development. Furthermore, some human activity that impoverishes ecosystems also contributes to increasing $CO_2$, emissions, notably by inhibiting their capacity to act as carbon sinks. Thus, the IPCC report for 2007 sets a figure of 20% for global $CO_2$ emissions resulting from deforestation. Recent work carried out under European programmes on permanent grassland – “GreenGrass” and “Carbo-Europe” - show that these grasslands are better able to capture $CO_2$ if they have a diversity of flora, which is encouraged by a more extensive type of management. Inversely, intensive management of the grassland impairs its role as a carbon sink (Soussana *et al.,* 2007).

# 3. Biodiversity at the heart of political and strategic decision-making

## 3.1. Biodiversity: a great number of socio-economic stakeholders

In the light of the vision proposed for biodiversity, every one of us is a stakeholder. Decision-makers at various levels - local to international - and different types - public, private, community or collective - are involved: governments, international organizations, local communities and native peoples, non-governmental organisations, academies, scientists, lawyers, the media, consumers, businesses, planners, developers, chambers of commerce… **All are involved in the decision-making processes that have a direct or indirect influence on biodiversity. The social and economic stakeholders are therefore extremely varied, and are not always fully aware of how they interact with biodiversity, either as beneficiaries or agents of interference.**

Institutional players in planning and politics, scientists, environmental protection groups and lawyers have long been direct traditional stakeholders. However, new players are emerging in the area of biodiversity, and with new roles and functions. A case in point is the private sector, which is now attempting to include biodiversity in business strategies, and even in accounting systems (Houdet, 2008). On a different scale, local and native populations and communities use biodiversity or their methods for managing it to back up their political, economic and even cultural claims. On another level, the finance sector is becoming involved in the area of compensation funds and trusteeships, particularly to finance protected areas for the long term.

## 3.2. Biodiversity at the heart of ethical and geo-economic issues

If you superimpose the global map of the concentration of biological wealth over that of economic wealth, you often see an inverse correlation between the two (UNEP-GRID Arendal, 2004). For example, Madagascar and Cameroon, two biodiversity hot spots, had a respective GDP per person of de 260 (2000) and 800 US dollars (2005).

In total, 1.3 billion people live in conditions of extreme poverty, generally in areas rich in biodiversity. **Local populations in developing countries mainly depend on the ecosystems in which they live for their food and their health; populations in rich countries make use of the ecosystems across the globe for goods (especially food and energy) and services, especially tourism** (UICN 2004, European Environment Agency, 2006). **This is what underlies the concept of the ecological footprint** which, although its calculation is contested, (Piquet *et al.,* 2007, Jolivet, 2008), enables us to report on the “burden” imposed on nature by a given population to support current levels of resources consumption and waste production (Wackernagel and Rees, 1996); and this extends way beyond their own borders for industrialised countries.

For local populations most often living under a regime where their rights of access and use are not assured, and under the pressure of investment benefiting industrialised countries, there is not much incentive to engage in a commitment to sustainable development of land and ecosystems.

Overseas communities are a special case: with the exception of Mayotte and Wallis and Futuna, the GDP per inhabitant is generally much greater than for neighbouring countries (Gargominy, 2003). For inhabited overseas communities, public transfers account for over 95% of the GDP, which has a strong effect on the relationship with biodiversity. Putting metal landing piers on Mayotte has had a negative impact on the life of the coral; waste outfalls on Guadeloupe are open to the skies and are located by the sea or, worse still, channelled to the edge of mangrove swamps in a central area of the National Park. A number of other regrettable examples could be cited here, which go against the general policy in favour of biodiversity. Furthermore, it was hardly logical for Guyana to clear land just to obtain title deeds to land if, at the same time, the State is calling for the conservation of ecosystems.

An important question for overseas communities is that of the preservation of the knowledge of customs of the local populations, currently being collected by businesses and NGOs. Being common to a group, local knowledge cannot be protected by a regime of private ownership: appellations of origin that protect a culture, landscape and techniques via a product are tools that can be used to resolve this problem.

A particular ethical and geo-economic issue is that of emerging nations such as Brazil, South Africa and even South East Asia, which have amazing levels of biodiversity. Their choice of embracing growth in order to achieve a standard of living comparable to that of "developed" countries threatens to have considerable impact on such biodiversity.

## 3.3. Biodiversity on the international political agenda

Started in 1980 with the "Global strategy for the conservation of living resources for sustainable development" commissioned from the IUCN by UNEP, political thinking on the **need for an integrated approach to conservation and biodiversity at all levels of decision-making and activity**, has not truly moved on since the Earth Summit in Rio de Janeiro in 1992, and the implementation of the Convention on Biological Diversity.

Ten years or so later, in 2002, an ambitious target was set by the Heads of State and Government of the signatories to the Convention on Biological Diversity, on the occasion of the 6th Conference for the Convention (La Haye) to: "***significantly reduce the loss of biodiversity between now and 2010 at a global, regional and national level in order to contribute towards the eradication of poverty for the benefit of all life on Earth"***. For its part, the European Union had taken a much more ambitious stance since 2001 – one of "*halting the loss of diversity between now and 2010*". Table II-4 summarises the important stages of this process.

"Arresting the loss of biodiversity and controlling climate change are the two most important issues facing the planet (…). In some ways, the loss of biodiversity is an even more serious threat, as the impoverishment of our ecosystems often reaches a point of no return, because a case of extinction cannot be reversed". These were the words of the European Commissioner for the Environment, Stavros Dimas, when introducing the annual "Green Week" in May 2006.

Aware of the necessity for national governance on biodiversity at the same level as for climate change, many countries are rallying to the idea, notably espoused by France, of a mechanism equivalent to the IPCC. Following the IMoSEB consultative process ("Towards an International Mechanism for Scientific Expertise on Biodiversity"), launched in 2005, **the principle of an *Intergovernmental Platform on Biodiversity and Ecosystem Services* (iPbes) was approved at the 9th Conférence of the Parties to the Convention on Biological Diversity** held in Bonn in May 2008.

### Table II-4: Recent political initiatives in favour of biodiversity

| | |
|---|---|
| **Global level** | |
| 6th Conference of the Parties to the Convention on Biological Diversity in La Haye, 7-19 April 2002 | Adoption of the Strategic Plan for the Convention on Biological Diversity (ruling VI/26), including the 2010 objective of "*significantly reducing the current rate of loss of biodiversity between now and 2010 at a global, regional and national level in order to contribute towards the eradication of poverty for the benefit of all life on Earth*". |
| World Summit on Sustainable Development in Johannesburg,<br>26 August - 4 September 2002 | Ratification of the 2010 objective aimed at significantly reducing the current rate of loss of biodiversity |
| 7th Conference of the Parties to the Convention on Biological Diversity in Kuala Lumpur,<br>9-27 February 2004 | Adoption framework (ruling VII/30) aimed at:<br>- enabling evaluation of progress towards 2010 objective and communications based on this evaluation<br>- encouraging consistency between work programmes under the Convention<br>- delivering a flexible framework where national and regional objectives can be included and indicators identified |
| **Pan-European level** | |
| 5th Pan-European Ministerial Conference in Kiev, 21-23 May 2003 | Ratification of a resolution to "Arrest the loss of biodiversity at all levels between now and 2010" via seven major areas of activity: forest and biodiversity, agriculture and biodiversity, pan-European ecological network, monitoring and indicators of biodiversity, invasive introduced species, financing biodiversity, public participation and raising awareness |
| **EU level** | |
| 6th Programme of Action for the Environment 2001 | "Nature and biodiversity" recognised as a priority theme in the general objective of "protecting, conserving, restoring and developing the function of natural systems, natural habitats, and wild flora and fauna with a view to arresting desertification and the loss of biodiversity (between now and 2010), including genetic diversity at EU and global level" |
| Gothenburg European Council , 15-16 June 2001 | Adoption of the EU Strategy for sustainable development, where one objective is to "manage natural resources more responsibly: protect and restore habitats and natural systems and arrest the loss of biodiversity between now and 2010" |
| Conference: "Secure conditions of life on Earth and biodiversity: reach the 2010 objective of the European Strategy for Biodiversity", Malahide, 25-27 May 2004 | Following a major consultation involving a number of stakeholders to revise the European Strategy (EC) for biodiversity and associated action plans - formulation of the "Malahide message" identifying the need for future action on transversal themes and in sectors with the greatest impact on biological diversity, in order to arrest the loss of biodiversity between now and 2010 |
| Brussels European Council , 28 June 2004 | Conclusions on "Arresting the loss of biodiversity between now and 2010" |
| Communication from the European Commission, May 2006 | Adoption of the Communication on "Arresting the loss of biodiversity between now and 2010 and beyond: supporting ecosystem services for the good of Mankind" plus a detailed biodiversity action plan from the EU |
| **National level** | |
| Several countries have included the 2010 objective in their National Biodiversity Strategy. France adopted a National Biodiversity Strategy in 2004, which is now supported by eleven industry sector action plans | |

*Source: CAS, Biodiversity Group*

## 3.4. Translating commitments made into working tools for the conservation of biodiversity

In support of these major political and strategic commitments, which are a reflection of a growing concern for the crisis facing biodiversity, we need to reinforce existing operational tools, replace them with new ones and above all structure them in a consistent strategic framework.

Like other countries, France has always had a varied range of legislative, territorial planning, land management and classification tools that can contribute more or less directly and restrictively to the protection of biodiversity (MEEDAT, 2008, Guillaume Grech MNHN-SPN, personal communication). Major examples are:

- **Regulation:** protection of species and reserves (9 National Parks, 1 Marine National Park, 30,909km$^2$ classified as nature reserves, 1,631km$^2$ classified as biotopes by Prefectural Decree, 8,000km$^2$ as classified sites, 17,000km$^2$ as registered sites).
- **Contractual protection:** regional reservation areas (71,773km$^2$), Natura 2000 (100,000 km$^2$ of land and sea), Pillar 2 of the Common Agricultural Policy (territorial agro-environmental measures, agro-environmental grassland payment) .
- **International classification:** biosphere reserves, Ramsar sites, World Heritage Sites.
- **Land management by the State:** *Conservatoire de l'espace littoral et des rivages* lacustres (1,200km$^2$, 1,000km of shoreline), public domain, *Office national des forêts* (18,000 km$^2$ in metropolitan areas); **by territorial communities:** management given over to the *Office national des forêts* (29,000 km$^2$ in metropolitan areas), sensitive wilderness areas; **by NGOs:** *Conservatoires régionaux d'espaces naturels* (1,400 km$^2$).
- **Environmental evaluation:** directive 85/337/CEE concerning the evaluation of the effects of certain public and private environmental projects, directive 2001/42 dated the 27$^{th}$ of June 2001, relating to the evaluation of the effects of certain public environmental plans and programmes; directive 2004/35 dated the 21$^{st}$ of April 2004 on environmental responsibility with regard to the prevention and repair of environmental damage.
- **Planning documents**: *directive territoriale d'aménagement* [land development directive] (DTA), *schéma de cohérence territorial* [land development plan] (SCOT), *plan local d'urbanisme* [local planning scheme] (PLU), especially classified woodland areas and directives for the protection and enhancement of landscapes.
- **Certification of production methods** (wood, appellation of controlled origin labels, organic agriculture).

However, given the extent of the erosion of biodiversity, a case by case use of these tools, without any overall integrated strategy for sustainable development, proved to be inadequate.

**In line with its commitment to the Convention on Biological Diversity, but under the more strict approach of the European Union, France adopted a National**

**Biodiversity Strategy in 2004, aimed at arresting the loss of biodiversity by 2010.** This National Strategy, with its eleven action plans[1], is a true "road map" from the French State for addressing the issue of the conservation of biodiversity in a much more integrated way, acting as it does at the level of sector policies.

**The provisions ratified under the Grenelle de l'environnement (new protected areas, including marine ones, acquisition of wetland areas, "green and blue open system", restoration of the ecological quality of water, improvement of knowledge and expertise) are intended to reinforce the mechanism for the conservation of biodiversity.** In terms of the huge uncertainties and upheavals principally linked to climate change, the development of a Green infrastructure, such as promoted by the European Union, or the "**green and blue open system**" for France, should encourage the linking of terrestrial and aquatic environments, at the same time contributing to the resilience of ecosystems and the maintenance of their capacity to deliver ecological services.

On a local scale, local communities, urged by new legal planning responsibilities, public opinion and the recent awareness of some elected representatives, are now committed to reviewing the issues and a structural outlook that includes biodiversity in regional development and planning policy: regional sustainable development schemes for their areas, biodiversity strategies at regional and department level, schemes for territorial consistency, Agenda 21 at local level and local planning schemes. Biodiversity is often seen as an essential element of attraction, particularly for the well-being of society, for certain economic activities benefiting from living resources and landscapes.

**The approach constituted by the economic evaluation of biodiversity, which is the main subject of this report, offers new leverage for intervention at an operational level, acting as a complement to those before it, in an attempt to arrest the loss of biodiversity.** This approach is intended to help direct decision-making for the use of natural resources and land use planning, by revealing the true value of biodiversity over the long term, when faced with economic interests that are often assessed over a shorter term.

[1] Action plans: natural heritage, agriculture, sea, planning, transport infrastructures, territories, international, overseas, forest, research, tourism (on-going).

# 4. The inclusion of the economic dimension in the approach to biodiversity

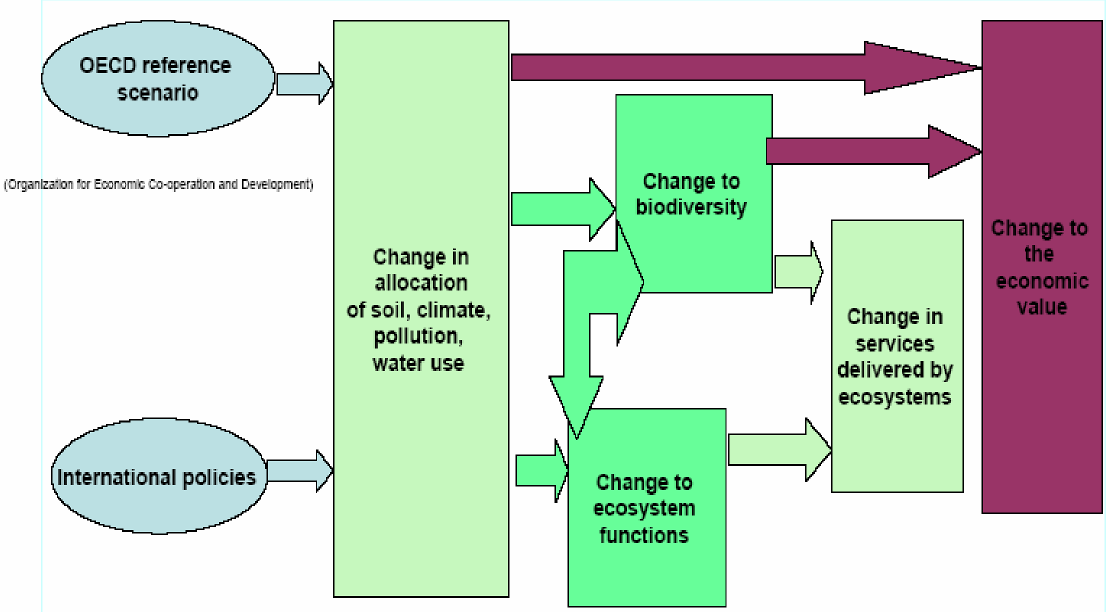
As well as the instruments cited above, a good number of routes for action are being explored at various levels. To begin with, it is important to make people aware of the cost of inaction by endeavouring to quantify that cost. It will then be appropriate to move towards establishing the true costs in terms of biodiversity as for other environmental areas. To do this, we will have to re-examine the public grants and taxes impinging on biodiversity. At a macro-economic level, a change to the national accounting system in order to include the value of biodiversity, and the implementation of payment mechanisms for maintaining and improving ecological services, are on the political agenda.

## 4.1. Evaluating the cost of inaction

The Stern Report, published in, 2007, evaluated the costs of inaction by 2050 in terms of climate change, and with resounding effect. **In 2007, after a meeting of Environment Ministers at the G8+5 in Potsdam presided over by Germany, the European Commission and the German Government decided to commission a similar type of study on the costs of inaction**, for the scenario where the 2010 objective for arresting the erosion of biodiversity would not be achieved (and we know that it won't). This study, entrusted to a team led by the economist Pavan Sudhev, and entitled "The Economics of Ecosystems and Biodiversity" (TEEB), is to conclude with the presentation of a report to the $10^{th}$ Conférence of the Parties to the Convention on Biological Diversity to be held in 2010. The group tasked with this study has published an interim report containing its initial instructive results.

To start with, the group developed **a logical framework for analysis** based on work by the OECD and the Millennium Ecosystem Assessment (figure II-2).

**Figure II-2 : Establishment of an analysis scenario for the cost of the loss of biodiversity**




*Source : The Economics of Ecosystems and Biodiversity, interim report, 2008*

Although they require development, initial evaluations carried out by Braat and ten Brink (2008) are extremely meaningful. Referring to the value of ecosystem services that might have benefited humans if there had been no losses to levels of biodiversity and it had been held at the respective levels of 2000 and 2010, the authors estimated what the annual monetary loss due to the loss of these services (table II-5) would represent in 2050; the impoverishment of ecological services would represent up to 7% of the global GDP in 2050, or **13,938 billion Euros per year.**

Table II-5: Value of losses associated with the impoverishment of ecosystems

| | Value of loss of ecosystem services per year in billions of Euros (10^9) | | | |
|---|---|---|---|---|
| | Full estimate | | Full estimate | |
| | Relating to 2000 | Relating to 2010 | Relating to 2000 | Relating to 2010 |
| | Billions of EUROS | Billions of EUROS | % GDP in 2050 | % GDP in 2050 |
| Natural areas | – 15,568 | – 12,703 | – 7.96% | – 6.50 % |
| Denuded natural areas | – 10 | – 6 | – 0.01 % | 0.00 % |
| Managed forest | 1,852 | 1,691 | 0.95 % | 0.87 % |
| Extensive agriculture | – 1,109 | – 819 | – 0.57 % | – 0.2 % |
| Intensive agriculture | 1,303 | 736 | 0.67 % | 0.38 % |
| Ligneous biofuels | 381 | 348 | 0.19 % | 0.18 % |
| Cultivated pasture | – 786 | – 1,181 | – 0.40 % | – 0.60 % |
| Artificial surfaces | 0 | 0 | 0.00 % | 0.00 % |
| Global total (terrestrial ecosystems) | **– 13,938** | **– 11,933** | **– 7.1 %** | **– 6.1 %** |

*Source: adapted from Braat and ten Brink (2008)*

## 4.2. Re-examination of the public incentives to impoverish biodiversity

**Today, we can consider that we do not count the true cost of using biodiversity. In other words, those who make use of biodiversity do not bear the direct costs; biodiversity payments do not cover the actual costs in terms of either the flow or renewal of stocks.** There are three reasons for this: the existence of public subsidies that lead to negative effects on biodiversity; the taxing of rural practices that are either only marginally viable or not at all at the same rates as for more viable activities; the continued negative tax costs in terms of biodiversity.

In practice, and in the current context of a market economy, it is partly because public assets, external effects and natural heritage are not evaluated appropriately that the costs of using or exploiting biodiversity are not taken into account in private decision-making processes. **Economic and fiscal instruments can therefore act to correct prices by factoring in the social and ecological costs that are currently seen by market mechanisms as external i.e. correcting the distortion between the cost to biodiversity and the cost to the producer /extractor.**

**Conversely, a number of direct or indirect public subsidies awarded for certain industry sectors in fact have a negative effect on biodiversity, in that they encourage the over-consumption or even destruction of it.** This does not mean that public subsidies need to be eradicated. They may be justified by other public policy considerations (social aspects, food autonomy, etc.). But a distinction needs to be made between two issues: on the one hand, the total cost of public support for an industry sector, and the nature of such support on the other. For example, if industry sector X in France receives €100M in public subsidies, the first task is to evaluate the necessity for such an amount of public support against the importance of that particular public policy, and the effectiveness of this support in industry sector X in achieving the relevant public policy objectives. If it is decided that €100M is still an appropriate level of public support for industry sector X, a second issue arises. If the proportion of public subsidies having a direct or indirect negative effect on biodiversity is estimated at 50%, whilst maintaining the same level of subsidies, it would then seem appropriate to find other ways of allocating the 50% in question in such a way as this proportion has a neutral effect in terms of biodiversity.

The phenomenon is a global one and, according to Norman Myers *et al.* (2007), **"pernicious subsidies", which are adversely affecting levels of biodiversity across the world estimated at 200 billion dollars/year, are actually up to ten times higher than the amount of expenditure devoted to nature conservation.** As early as 1998, the 4th Conference of Parties to the Convention on Biological Diversity (CDB) was encouraging signatories to identify any pernicious incentives and review ways of eliminating them or reducing the effects on biodiversity. Resolving the problem of subsidiaries with adverse affects is at the core of the programme of work on biodiversity incentive measures implemented by the CBD.

**Two industry sector policies in force across Europe – for agriculture and fishing – illustrate this problem in particular.**

As emphasised by the recent NIAR expert report, "Agriculture and biodiversity - promoting synergies", **direct agricultural aid paid to French producers under the first Pillar of the Common Agricultural Policy** (support for markets and revenues) is much greater than that granted under the rural development policy of the second Pillar (11 billion as against 1.5 billion Euros). Even though CAP reform has been under way since 1992, with a view to the gradual shift of budget resources from the first to the second Pillar, and with the priority moving towards land and the environment, it cannot be ignored that this shift is only a modest one. But it is hard to know what the "optimal" distribution of funds between the two Pillars might be, especially given the difficulty of measuring the value of the environment and the services it provides. However, the introduction of the principle of eco-conditionality to the first Pillar offers a chance to work additionally on measures that favour biodiversity.

**Table II-6: Distribution of agricultural support in 2007**

| | Surface (millions of hectares) | Proportion of territory | Proportion of French SAU | Aid (billions of Euros)* |
|---|---|---|---|---|
| First Pillar (agricultural markets and revenues) | | | | 11 |
| SCOP (conditionality) | 13 | 24 % | 40 % | 6.6 |
| inc non industrial set aside | 1.2 | 2.2 % | 4 % | na |
| Second Pillar (rural development) | | | | **1.51** |
| Total MAEs | 6.9 | 12.7 % | 30 % | 0.6 |
| Of which ABs | 0.6 | 1.1 % | 2 % | na |
| Of which grassland payment | 3.2 | 5.8 % | 10 % | 0.21 |
| Natura 2000 (1,705 sites) | 6.8 | 12.4 % | 10 % | 0.02 |
| Regional reservation areas (45) | 7 | 12.7 % | 10 % | na |

* Direct aid paid under the Agricultural Policy; na = not available.
*Source: MEEDAT and MER*

The lack of consistency between policies has direct repercussions for the behaviour of farmers with regard to biodiversity. Thus, a farmer wanting to adopt a measure favourable to the environment (for example, an agro-environmental measure (MAE) favourable to biodiversity) might not go ahead, as it would force him to relinquish all or part of other types of direct aid paid in relation to specific objectives: this would apply if the MAE requires a reduction in the number of animals per surface unit, when another type of aid would be increased to match a higher number of animals (Le Roux *et al.*, 2008).

**Over-fishing, which occurs throughout European waters, is leaving fishing companies in serious difficulties.** Policies implemented since the 1970s and the associated subsidies under the Common Fishing Policy are contributing to both over-fishing and economic impoverishment, in the belief that modernising the fleets can solve the problem, when in fact they have two major impacts on renewable resources:

- Modernised fleets increase the pressure on already fragile ecosystems, in time causing problems for the companies themselves;
- Modernised fleets replace labour with assets (technology).

According to researchers at the University of British Columbia (Jacquet and Pauly, 2008), la traditional fishing practices are at a double disadvantage: due to eco-labelling and classification initiatives on the one hand, as they essentially apply to large scale fisheries, and by politically ill-conceived fuel subsidies on the other. Thus, of the 30 to 34 billion dollars (22 to 25 billion Euros) spent on aid every year for this industry sector, only a fifth reportedly goes to small-size fisheries (i.e. operated on ships less than 15m long). However, according to the same authors, traditional fisheries still in fact catch just as many fish for human consumption as industrial fisheries, as they are much more selective. Whilst being less destructive in terms of marine ecosystems, they use eight times less fuel, and provide many more jobs.

**Referring to estimates from the World Bank, the Senator Le-Cléach report (2008) offers some startling figures: the loss of earnings compared to the optimum level, if fishing was managed properly, would be 57 billion dollars, set against fishing revenues of 85 billion dollars. This would make the loss of income between 1974 and 2004 around 2,200 billion dollars**, and these were considered to be conservative figures. On a global scale, annual losses would amount to 64% of the value of the catch and 71% of the value of the fish exchanged at international level.

Managing access to resources is the *sine qua non* condition for any sustainable management system for eco-systems and protected areas. The solution runs from management of access to resources right up to instruments for economic management (licences, operating rights or negotiable access rights).

## 4.3. Proposing major tax changes

The Millennium Ecosystem Assessment (MEA) considers that there are four types of capital: manufacturing capital, social capital, human capital and natural capital. Most of the current taxation in the world today relates to manufacturing and human capital (through work). **MEA experts believe that the current environmental crisis is mainly the result of the type of regulation that makes us consider that natural resources (especially renewable ones) and ecological services come free of charge. They**

**believe that it is necessary to start planning now to replace the tax on manufacturing and human capital with a tax on the consumption of natural resources.**

In terms of biodiversity, setting a tax on our heritage is important, because most of the damage done to natural environments is done by the bias un-built land, itself subject to tax.

From the perspective of biodiversity, we can formulate **four basic criticisms against the French tax system** (Sainteny, 1993).

1. **The equal level of taxation on assets, where the level of yield is very different, especially in terms of fixed as opposed to non-fixed assets.** Of little effect on a holdings consisting of securities (long-term yield hovering around a figure of 7%), but of greater importance for rented property (long-term yield hovering around a figure of 4-5%), a tax of 1.5% or 2% becomes confiscatory, and acts as a direct incentive to the sale, development or artificialisation of holdings consisting of un-built land assets (wilderness areas, forests, land in the possession of rural owners not being used etc.), where the yield level is nearly always less than 2%. In the first two instances, tax will be paid with the revenue from the asset, but this seems impossible in the final instance.
2. **The same equal level of taxation on very different types of un-built land (whether vanishing areas or otherwise) where very different methods of using it are employed (intensive, extensive or biological agricultural use) results on the one hand, in the penalisation of non-intensive use** - less productive, but requiring greater surface area, and, on the other, intensifying the yield from the least viable un-built land to compensate for the new tax with new revenues.
3. **In general, un-built land attracts higher taxes in France than in comparable countries. This is the case compared to Germany, Spain, Great Britain and the United States.**
4. **In the case of France, the French Council of Taxes has shown that, over a period of 29 years, the net rate of return of an agricultural property in terms of indirect value was negative in all cases concerned** (French Council of Taxes, 1986). The issue regarding un-built, non-agricultural land is without doubt a more delicate one. On the one hand, it doesn't produce any revenue directly, and is therefore unable to pay the related property tax from any earned revenue. On the other hand, it is often richer in biodiversity, producing goods and ecosystem services that currently have no economic value, but which are very useful to society, and therefore the economy. As currently evaluated in economic terms, the viability of these areas, many of which should probably be preserved from a biodiversity point of view, is therefore even more in the negative, which in turn increases the pressure for them to be developed. Furthermore, an equal level of tax on such landed property offering weaker returns in fact leads to relatively higher taxation for lower net returns, and therefore to higher taxes on wilderness areas with higher levels of biodiversity. **In such cases, taxation, in penalising rich ecosystems to a relatively higher degree, encourages them to be developed, and drives society and the economy towards impoverishment by robbing them of an important part of**

**the ecosystem goods and services that were provided to them free of charge by these same wilderness areas.**

In order to meet the relatively high levels of tax on a un-built land with a gross yield that is already very low, its owner must therefore either increase the gross yield of his asset (more intensive use, change of use, development, etc.) or eliminate it, meaning nature in this case.

**If we want to increase the return on un-built land without necessarily having to resort to development, urbanisation or intensified use, it seems necessary to lighten the burden of tax on it.**

**Furthermore, a certain amount of tax expenditure directly or indirectly encourages the impoverishment of biodiversity** (see table II-7). This tax expenditure is a triple negative: for biodiversity by its very nature, for the State budget, because it represents a loss of tax revenue, and for the economy and society as a whole, because it does not recognise true costs, which is an essential element of consumer choice. Again, this does not mean that they must all be eliminated, or that the industry sectors involved should necessarily see a reduction in the public support from which they benefit. But these costs still need to be reviewed in terms of application and reduced in number and level.

Table II-7: Types of tax expenditure for review
in relation to their impact on biodiversity

| Type of tax measure | Reference | Tax in question | 2007 PLF budget impact (in €M) | Potential negative effects on biodiversity and true costs |
|---|---|---|---|---|
| Rate of 5.5% applicable to ingredients of animal feed and certain products/fertilizers for agricultural use | Article 278a -4 and 5 of the French Tax Code | VAT | 50 | Underestimates the cost of phytosanitary products, and can therefore encourage use, whereas the Grenelle requests that they be reduced |
| Business tax exemption for assets of any kind used for irrigation to 9/10ths of their capacity | Article 1469 of the French Tax Code | TP (French public works) | | Underestimates the cost of irrigation, and can therefore encourage use thereof and the increase in crops that are high consumers of water and relatively poor in biodiversity |
| Hedge removal, filling ditches, drainage… deductable from tax revenue for rural property | LFI 2006 (French budget law) | IR | | Encourages removal of important structural elements of certain ecosystems and landscapes (enclosures, wetlands) |
| Reduced rate of TIPP (French national tax on petroleum products) applicable to domestic fuel used as diesel fuel | Articles 265 and 265 B of the French Customs Code | TIPP | 1,450 (950M Euros in 2009) | Provision historically implemented to mechanise agriculture in France, which has now been widely accomplished. Leads to under-estimating the cost of using additives (fertilizer, phytosanitary products…) and relatively disadvantages organic or extensive agriculture, which are less mechanised and use fewer additives |
| Business tax exemption for mine concession holders, mining permits and people exploring fields for combustible oil and gas | Article 1463 A of the French Tax | TP | | The impacts of offshore oil and gas oil field exploration on the aquatic environment are being increasingly documented |

| Type of tax measure | Reference | Tax in question | 2007 PLF budget impact (in €M) | Potential negative effects on biodiversity and true costs |
|---|---|---|---|---|
| | Code | | | |
| Deduction from global revenue of paid subscriptions to capital with companies approved for financing traditional fishing methods (SOFIPECHE) | Article 163 duovicies, 238 a HO, 238 a HP of the French Tax Code | IR (French revenue tax) | | Provision that encourages over-capacity of fishing fleets and therefore too great a pressure on fishing stocks in relation to their actual status |
| VAT exemption on delivery, repair, maintenance, chartering and rental operations relating to boats used for professional sea fishing and for deliveries of goods for supplies to these boats | Article 262-II- 2 and 6 of the French Tax Code | VAT | | *Idem* |
| 50% reduction on the taxable income of young fishermen starting between 01/01/1997 and 31/12/2010 | 44 nonies of the French Tax Code | IR | | *Idem* |
| Exemption of operations carried out by fishermen and fishing boat owners, apart from fresh water fishermen, with regard to the sale of fishing products (fresh or chilled fish, crustaceans and shells) | 261- 2-4 of the French Tax Code | VAT | 60 | *Idem* |
| Exceptional depreciation of 50% of sums paid for subscriptions to capital with companies approved for financing for financing traditional fishing methods (SOFIPECHE) | 217 decies, 238 a HP, 238 a HO of the French Tax Code | IS (French revenue tax) | | *Idem* |
| TIPP exemption for certain fuels used for navigation, notably for fishing in Community waters | Article 265a of the French Customs Code | TIPP | 210 | *Idem*<br>Furthermore, a provision that encourages French fleets to use machinery (trawling equipment) that causes both a serious impoverishment of the marine environment itself (sea bed) and significantly higher energy use than for other fishing methods |

**A number of provisions adopted between 2005 and 2007 have improved the tax status of wilderness areas:**

- Since 2005, according to the Law on the development of rural territories, Natura 2000 sites are 100% exempt from the community portion of the French un-built property tax (TFNB), and certain wetlands are 50% exempt, subject to committing to a management plan for five years.
- Protected wilderness areas are now subject to the same regime as rural property on a long-term lease in terms of free rights of charge.
- The restoration and maintenance costs for these same areas are now deductible from land revenue.

- Since the National Parks Law was passed in April 2006, central areas of National Parks located in overseas departments are exempt from the TFNB [French un-built property tax] for five years.

Furthermore, as part of the *dotation globale de fonctionnement* [French State Budget Grant] (DGF), the National Parks Law of April 2006 instituted a grant to be paid to communities whose territories are totally or partly within a National Park. This grant relates to the proportion of the surface area of the community included within the park, and is doubled for calculating the grant if this area exceeds 5,000km². It changes every year in line with the *dotation globale de fonctionnement*. This measure is an important step in creating a sense of solidarity for communities that have natural heritage in need of protection.

**Nevertheless, the tax status of our natural heritage is still not on a level with that of our cultural heritage. For this to happen, the costs of restoring and maintaining protected wilderness areas would need to be deductible from global revenue (with a ceiling) and not just from land asset, revenue, and for wetlands to also be exempt from the TFNB [French un-built property tax].**

A number of other measures were proposed by Senators P. Laffitte and C. Saunier in 2007:

- Review the tax incentives for the development of natural environments,
- Reduce the tax pressure on natural environments. For example, it should be possible to apply full or partial exemptions to wetlands and wilderness areas with strict protection status, areas now used for organic farming and natural grassland.
- Encourage the use of local taxation systems to slow down urban sprawl.
- Provide incentives for restoring wilderness areas through applying a tax on revenue and capital.
- Use local community finance grants in such a way as to encourage biodiversity. The inclusion of a category for biodiversity for calculating the *dotation globale de fonctionnement* [French State Budget Grant] (DGF) – proposed by working group 2 of the *Grenelle de l'environnement* – is a move in this direction. But we need to go further, and, as in the case of the National Parks Law referred to above, consider the setting up of protected areas by local communities as an investment, just as we do the renovation of a school or building a road; this is why the *dotation globale de fonctionnement* [French State Budget Grant] (DGI) should include a "biodiversity" category.

The group generally considers that, in terms of biodiversity, the use of economic and tax instruments does not have to be prioritised towards yield and finance goals, but more towards incentives, internalisation of costs and true pricing.

## 4.4. Revising the National Accounting system to include the value of ecosystems

The United Nations has produced a theoretical framework for drawing up integrated economic and environmental accounts in the form of the SEEA3. Environmental accounts are currently quite well integrated within the French national accounting system (SCN), and mainly cover issues linked to production and consumption pressures. **Under the SEEA review, which is planned for 2012, the intention is to include accounts for ecosystems in order to measure the impact of such pressures on the working of ecosystems themselves, and the associated consequences for the services that they deliver to the economy and to the general well-being of humanity.** A number of basic questions need to be answered with regard to the sustainability of the interaction between nature and the economy (Weber, 2008):

- Are our natural renewable assets (ecosystems, with their functions and services) being maintained across time?
- Is the full cost of maintaining and restoring natural assets covered by the current cost of the goods and services?
- Does the price of imported products cover the full costs of maintaining and restoring ecosystems in the country of origin?
- Does the overall requirement for the goods and services delivered by the economy and ecosystem services used free of charge by households, both individually and collectively, increase over time?

This "macro" approach is a response to the need to establish aggregated operational values, as expressed through the issues raised at a political level in recent few months:

- The 2007 "Stern" report on the cost of inaction with regard to climate change,
- The dG8+5 Potsdam initiative (2007) for a report on the costs of inaction with regard to the loss of biodiversity, entitled "TEEB: The Economics of Ecosystems and Biodiversity",
- The European Conference entitled "Beyond the GDP" in November 2007,
- The increased in the number of initiatives for evaluating and accounting for ecosystems (Green Accounting for Indian States Project, Eureka!-Europe: national programmes being prepared in the UK, France and Spain etc.),
- The World Bank programme for calculating the "true net saving" (now "adjusted saving"),
- The "Green economy initiative", recently launched by UNEP (see below),
- The human development index (HDI), created by UNEP in 1990,
- Work in France led by the Commission for measuring economic performance and social progress (Sen-Stiglitz Commission).

**Finally, raising the issue of international payments for ecosystem services, in agriculture for example (FAO, 2008), where accounting is as necessary as for the Kyoto Protocol, is a considerable challenge now facing the SEEA.**

The European Environment Agency is actively engaged in carrying out work on a methodology for ecosystem accounting (Weber *et al.,* 2008), and is participating in this capacity in one of the five sections of the second phase of the TEEB, which relates to "The scientific bases: a framework for evaluation, economic evaluation methodology and costs analysis". Results from this work are expected in 2010.

### 4.5. Integrating biodiversity into business accounting and strategy

The Convention on Biological Diversity explicitly acknowledges, particularly in articles 10 and 16, the importance of engaging the business world in the conservation of biodiversity. Started in 2005, and with strong support from the European Commission, a specific dynamic approach was adopted at the Conference of the Parties to the Convention held in Curitiba, Brazil in March 2006, under the heading of "Business and Biodiversity", and then at the EU Lisbon Conference in 2007, presided over by Portugal (same heading).

This initiative is based on organising regular meetings around this topic, and calls for the adoption of "best practices" to minimise the impact of businesses on biodiversity and promote its conservation. It starts from the premise that "***the variability and uncertainty associated with biodiversity and ecosystem services are both risk factors and opportunities for businesses, particularly in terms of the supply of raw materials, reputation, capital costs and regulation.*** *In order to ensure the continuation of the organisation, certain methodologies will enable us to prioritise issues for the purposes of decision-making and taking action and to move towards a better control of the impacts in question; by factoring biodiversity into the economy, and giving it a fair price"*(Houdet, 2008).

Aligned with this, the French Biodiversity Institute (IFB) and the Orée Association decided at the end of 2005 to set up a working group on biodiversity, bringing businesses, scientists, associations and communities together.

**The aim of the group is to research ways in which biodiversity can actually be a motive force for development and economic activity a way of preserving or increasing biodiversity.**

Initial work started with establishing an **indicator for the interdependent relationship between a business and biodiversity (IIEB), a self-assessment tool that should enable businesses to identify the direct and indirect interactions they have with the living world.** This approach enabled 25 businesses involved in this evaluation process to take the concept of biodiversity fully on board and position themselves in relation to certain criteria chosen for revealing the most information, and to set the groundwork for the implementation of strategic action.

Starting from the premise that business accounting systems are not designed to evaluate and monitor relations between business and biodiversity, the next step for the work of the group is to establish a methodology for drawing up a Business ***biodiversity balance sheet,*** which would be the biodiversity element of the "Carbon balance sheet" - staying within **business language** - one of *costs* and *benefits*.

### 4.6. The "Green economy initiative"

On the 22nd of October 2008, UNEP and top economists launched the **"Green economy initiative", which is aimed at profiting from historical circumstances to deliver the economy of tomorrow today.** Mobilising the global economy and redirecting it towards investing in clean technology and natural infrastructures, such as land and forests, is seen as the best bet for achieving real growth, combating climate change and creating jobs in the 21st century.

There are three key aspects to the green economy initiative: evaluating and integrating the national and international potential of nature, creating green jobs and establishing market policies, instruments et signals that will accelerate the transition towards a green economy.

This strategy, built on the results of the TEEB is also linked to the green job initiatives of UNEP, the International Employment Organization, the International Trade Union Confederation and the International Organisation of Employers.

The Green economy imitative will rely heavily on the considerable amount of work already done by UNEP, the United Nations system and other organisations, ranging from the impact and potential of fishing, fuel and other subsidies to innovative market mechanisms and financial products that area already making a start with the transition.

## Conclusions

In this chapter, we have attempted to demonstrate the extent but also the complexity of the socio-economic and political issues associated with biodiversity, particularly due to the large number of players and the connections between the levels at which these players interact with biodiversity.

**Faced with a historic biodiversity crisis, at the dawn of a possible sixth extinction and under growing human pressure, traditional approaches to the conservation of biodiversity have proved highly inadequate in terms of halting and reversing the trend.** France itself has huge responsibility in terms of conservation issues, given that, including its overseas territories, it is home to over a third of the world's biodiversity.

There is now a great awareness of the need for action, and biodiversity is now on the international political agenda, with a commitment to eradicate the loss of biodiversity by 2010.

This general awareness has been raised by revealing – notably through the *Millennium Ecosystem Assessment* – the extreme level of dependency of human activity on biodiversity, far beyond the market products from our current levels of biodiversity (food, medicines, fibres, wood, etc..), whether this be fossil biodiversity (fuels, materials and minerals) or support, regulatory, cultural or inspiration-related services (the basis for bionics). The great uncertainties of climate change should serve to remind us that biodiversity **is a guarantee of "life insurance for life itself".**

The evaluation of the socio-economic issues of biodiversity leads us to revisit in detail the notion of the externality of costs for programmes, projects and activities over the long term, and to think about the various areas where we could take action to promote biodiversity:

- At a "micro-economic level", taxation **has proved to be a powerful lever for accounting for biodiversity and the services it delivers**, both by reviewing the subsidies and tax costs that have a negative effect on biodiversity, and by setting up a system of incentives. **Including biodiversity in business accounting and strategy** also applies at this micro level.

- At a macro-economic level, , the initiative to **revise the National Accounting System** by including a "value" for biodiversity and the services it delivers is allied to a number of emerging approaches aimed at **the establishment of a market for the payment of ecosystem services**, or, most recently, the "Green economy initiative".

There is now a real, perceptible momentum for making economics part of approaches to the conservation of biodiversity. However, there is still major uncertainly around how to achieve this. We still need to acquire considerable knowledge of the ecological functions that are linked to biodiversity, and of the limits of resilience of ecosystems to the effects of growing pressure. Methodologies need to be fine-tuned to establish indicators for biodiversity and the integrity of ecosystems relevant at various scales (see chapter IV). Traditional economic evaluation methods need to be tested in relation to things that are often hard to quantify, such as regulatory or cultural services related to ecosystems. And this needs to be carried out in a sufficiently robust way, so as to run hand in hand with recent developments in the Law towards the concepts of exchange and compensation (see chapter III).

# Bibliographic References

Babin D., Thibon M., Larigauderie A., Guinard S., Monfreda C. et Brels S. (2008), *Strengthening the science-policy interface on biodiversity. Results of the consultative process towards an IMoSEB*, IMoSEB / IFB / Cirad, 128 p.

Balmford A., Rodrigues A.S.L. *et al.* (2008), *The Economics of Biodiversity and Ecosystems: Scoping the Science*, Cambridge, UK, European Commission: 297.

Benyus J. M. (2002) (réédition d'un ouvrage de 1997) (New Edition of a 1997Book), *Biomimicry, Innovation Inspired by Nature*, Harper Collins, New York, 301 p.

Blondel J. (2004), « Impact du réchauffement climatique sur les oiseaux: ajustements phénotypiques ou réponses adaptatives ? », *in Actes des Deuxièmes Journées de l'Institut français de la biodiversité*, Marseille, 25-28 mai 2004. (Impact of Global Warming on Birds: phenotypic Adjustments or adaptive Responses in Proceedings of Day #2 Discussions at the French Biodiversity Institute, Marseille, 2004 May 25th-28th)

Braat L. et ten Brink P. (eds) (2008), *The Cost of Policy Inaction: The Case of not Meeting the 2010 Target*, Study for the European Commission, DG Environment in support to the TEEB First phase.

Burel F. et Baudry J. (2003), *Landscape Ecology, Concepts, Methods, and Applications*, Science Publisher, New Hampshire, 378 pages.

Centre de recherches de climatologie (CRC) et Chaire UNESCO Vins et Culture (2007), Actes du colloque international « Réchauffement climatique, quels impacts probables sur les vignobles ? », Dijon, 28-30 mars 2007. (Climatology Research Center (CRC) and UNESCO Chair on Wines and Farming (2007), Proceedings of International Symposium Addressing " Global Warming, What Predictable Impacts on Vineyards ?")

Centre thématique européen sur la diversité biologique (CTE/BD) (2008a), Liste de référence des espèces et habitats de la directive Habitats par région biogéographique, http://biodiversity.eionet.europa.eu/ (accédé le 10/01/09) (European Thematic Center for Biological Diversity (CTE/BD) (2008a). Reference List of Species and Habitats in the Habitat Directive per Geographic Region). (Accessed on 09/10/01)

Centre thématique européen sur la diversité biologique (CTE/BD) (2008b) (European Thematic Center for Biological Diversity (CTE/BD) (2008b), *Article 17 Web Tool*, « Biogeographical assessments at MS level », http://biodiversity.eionet.europa.eu/article17 (accédé le 10/01/09).

Chevassus-au-Louis B. et Griffon M. (2007), « La nouvelle modernité: une agriculture productive à haute valeur écologique »,(New Modernity: a Productive Agriculture Featuring High Ecological value) *DEMETER 2008*, 7-48.

Cleach M. P. (2008), *L'apport de la recherche à l'évaluation des ressources halieutiques et à la gestion des pêches*, Office parlementaire d'évaluation des choix scientifiques et technologiques (Contribution of Research to the Assessment of Halieutic Resources and to the Management of Fishing Activities, Parliamentary Office in charge of Assessing Scientific and Technological Choices) , 174 p. Collectif d'experts (Experts Group) (2003), *Stratégie nationale pour la biodiversité, état des lieux (National Strategy for Biodiversity, Current Situation (2003)*, www.ecologie.gouv.fr/IMG/pdf/etat_des_lieux_-_decembre_2003.pdf.

Collectif Science & Décision (Science&Decision Group)(2007), *La Biodiversité dans les zones rurales (Biodiversity in Rural Zones)*, université d'Evry-Val-d'Essonne (Evry-Val-d'Essonne University), CNRS, www.science-decision.fr/cgi-bin/topic.php?topic=RUR&chapter=0.

Commission européenne (European Commission)(2008), *L'économie des écosystèmes et de la biodiversité (Ecosystems and Biodiversity Economics)* , Luxembourg, Office des publications officielles des Communautés européennes (Office for Official Publications of the European Communities), 64 p.

CSPNB (Conseil scientifique du patrimoine naturel et de la biodiversité) (Scientific Council for Natural Patrimony and Biodiversity) (2007), *La biodiversité à travers des exemples (Biodiversity Through Examples)*, MEDD/D4E, 104 p., www.ecologie.gouv.fr/-CSPNB-html.

Cury P. et Miserey Y. (2008), *Une Mer sans poissons (A Fishless Sea)*, Paris, Calman-Lévy, 279 p.

EFTEC (2005), *The Economic, Social and Ecological Value of Ecosystem Services: A Literature Review*, Report to DEFRA, Londres, 42 p.

European Commission (2008), *Biodiversity Communication Action Plan COM(2006)216: Communication from the Commission - Halting the loss of biodiversity by 2010 – and beyond – Sustaining ecosystem services for human well-being*, Brussels.

European Environment Agency (2008), *Ecosystems Services – Accounting for what Matters*.

European Environment Agency (2006), *Progress Towards halTing the Loss of Biodiversity by 2010*, EEA Report No 5/2006, Luxembourg, Office for Official Publications of the European Communities.

FAO (2007), *La situation mondiale de l'alimentation et de l'agriculture 2007. Payer les agriculteurs pour les services environnementaux (World Situation Regarding Food and Agriculture. Paying Farmers for the Environmental Services they Provide)*, Rome, 240 p.

Feldmann P. (2007), « Une floraison précoce d'*Ophrys* dans l'Hérault: un effet du changement climatique ? », (The Early Blossoming of Ophrys in Hérault: A Consequence of Climate Change ?)*L'Orchidophile*, N° 38 (2), p. 111-116.

Gallai N., Salles J. M., Settele J et Vayssière B (2008), « Economic valuation of the vulnerability of world agriculture confronted with pollinator decline », *Ecological Economics*, doi:10.1016/j.ecolecon.2008.06.014.

García de Cortázar Atauri I., Brisson N., Jacquet O., Payan J. C., Claverie M., Salançon E., Fournioux J. C., Monamy C., le Sueur D., Barbeau G., Bottois N., Dumot V., Gaudillere J. P., Goutouly J. P., Agut C., Rodriguez-Lovelle B., Ramel J. P., Huard F., Ripoche D., Launay M. et Seguin B. (2007), « Évaluation des impacts du changement climatique dans les différents vignobles de France à l'aide du modèle STICS-vigne » (Evaluating the Impact of Climate Change on the Different Vineyards in France using the STICS-Vineyard model), *Actes du Congrès Climat et Vigne*, avril 2007 (Proceedings of the Climate and Vineyards Congress, April 2007,Zaragoza, 8 p.

Gargominy O. (éd.) (2003), *Biodiversité et conservation dans les collectivités françaises d'Outre-mer*, (*Biodiversity and Conservation in the French Overseas Collectivities*) Collection Planète Nature, Comité français pour l'UICN (Planet Nature Collection, French Committee for IUCN), Paris, 246 p.

Griffon M. (2006), *Nourrir la planète (Feeding the Planet)*, Paris, Odile Jacob, 456 p.

Guillot A. et Meyer J.-A. (2008), *La bionique, quand la science imite la nature (Bionics, when Science Imitates Nature*) Dunod, Collection UniverSciences, 248 p.

Houdet J. (2008), *Intégrer la biodiversité dans les stratégies des entreprises: le bilan biodiversité des organisations*, (Integrating Biodiversity into Firms Strategy: Biodiversity Achievements in Organisations) (Fondation pour la recherche sur la biodiversité, Entreprises, territoires et environnement. (*Foundation for Research Applied to Biodiversity, Firms, Territories and Environment)*

IFEN (2006), « L'Environnement en France » (Environment in France), Édition 2006, *Les Synthèses de l'IFEN*, IFEN, Orléans, 471 p. + annexes.

INPN-MNHN (2009), Site de l'Inventaire national du patrimoine naturel, (Site of the National Inventory for Natural Patrimony)http://inpn.mnhn.fr/isb/index.jsp, (accédé le 3 février 2009).

Jacquet J. et Pauly D. (2008), « Funding priorities: big barriers to small-scale fisheries », *Conservation Biology*, Volume 22, Issue 4, p. 832-835.

Jolivet P. (2008), « La double erreur de l'empreinte écologique » ("The Twofold Mistake of Ecological Footprint"), *Les Échos*, 26 novembre.

Klein A. M., Vaissière B. E., Steffan-Dewenter I., Cunningham S. A., Claire K. et Tscharntke T. (2007), « Importance of pollinators in changing landscapes for world crops », *Proceedings of the Royal Society of London*, volume 274, n° 1608, 7 février.

Körner C. (2005), « Biodiversity and atmospheric change », *International Conference, Biodiversity, Science and Governance,* Paris, janvier.

Laffitte P. et Saunier C. (2007), *Rapport sur les apports de la science et de la technologie au développement durable (Report on the Contribution of Science and Technology to Sustainable Development)*, Tome II: « La biodiversité: l'autre choc ? l'autre chance ? » (Biodiversity: the Other Shock, the Other Opportunity ?), Office parlementaire d'évaluation des choix scientifiques et technologiques. (Parliamentary Office for Assessing Scientific and Technological Choices)

Lepart J. (2005), « Diversité et fonctionnement des écosystèmes et des paysages » ("Diversity and Functioning of Ecosystems and Landscapes"), *in Les biodiversités - Objets, théories, pratiques*, (*Objects, Theories, Practices)* P. Marty, F.-D. Vivien, J. Lepart, R . Larrere (éds), CNRS Éditions, novembre.

Le Roux X., Barbault R., Baudry J., Burel F., Doussan I, Garnier E., Herzog F., Lavorel S., Lifran R., Roger-Estrade J., Sarthou J. P. et Trometter M. (éds) (2008), « Agriculture et biodiversité. Valoriser les synergies. Expertise scientifique collective »,("Agriculture and Biodiversity. Valuing Synergies. Collective Scientific Expertise") *synthèse du rapport INRA (Executive Summary of the INRA Report)* (France), 113 p.

Levrel H. (2007), *Étude de faisabilité pour la réalisation d'un « Millennium Ecosystem Assessment en France », (Feasibility Study Concerning the Making of a Millenium Ecosystem Assessment in France)* Rapport interne au ministère de l'Écologie et du Développement durable (Report Internal to the Ecology and Sustainable Development Ministry), MNHN, Paris.

Mander U., Wiggering H. et Helming K. (eds) (2007), *Multifunctional Land Use, Meeting Future Demands for Landscape Goods and Services*, Springer-Verlag Berlin Heidelberg, 421 p.

Marangon F. et Visintin F. (2007), « Rural landscape valuation in a cross-border region », *Cahiers d'économie et sociologie rurales*, n° 84-85.

Marbaix P. et van Ypersele J.-P. (2005), *Impacts of Climate Change in Belgium*, Report commissioned by Greenpeace, université catholique de Louvain, Belgique.

McNeill C. et Shei P. (2002), *A Framework for Action on Biodiversity and Ecosystem Management*, Water-Energy-Health-Biodiversity (WEHAP) Working Group, World Summit on Sustainable Developpement. Johannesburg, Afrique du Sud, 26 août-4 septembre 2002, Nations unies, New York.

MEEDAT (2009), Site Internet section « Natura 2000, Espaces protégés, Sites classés, Paysages (Protected Areas, Sites with a Preservation Order, Landscapes) ». www.ecologie.gouv.fr/-Biodiversite-et-paysages-.html (accédé le 26/01/2009).

MEEDAT (2008), *L'Outre-mer français, un patrimoine naturel d'exception,(French Overseas Territories, an Exceptional Natural Patrimony*) Rapport édité par le MEEDAT, 34 p. www.ecologie.gouv.fr/IMG/pdf/3%20-%20brochure%20francais.pdf (accédé le 10/01/09).

Millennium Ecosystem Assessment (2005a), *Ecosystems and Human Well-being: A Framework for Assessment*, Island Press, 201 p.

Millennium Ecosystem Assessment (2005b), *Ecosystems and Human Well-being: Biodiversity Synthesis*, World Resources Institute, Washington DC., 86 p.

Molyneux D. H., Ostfeld R. S. *et al.* (2008), *Ecosystem Disturbance, Biodiversity Loss and Human Infectious Disease. Sustaining Life. How Human Health Depends on Biodiversity*, E.C. a. A. Berstein. New York, Oxford University Press: 287-323.

Myers N. (Lead Author), Golubiewski N. E. et Cleveland C. J. (Topic Editors) (2007), « Perverse subsidies », *in Encyclopedia of Earth*, C. J. Cleveland ed. (Washington D.C.: Environmental Information Coalition, National Council for Science and the Environment) [Published in the *Encyclopedia of Earth*, August 9; www.eoearth.org/article/Perverse_subsidies (accédé le 11/01/09).

Newman D. et Cragg G. (2007), « Natural products as sources of new drugs over the last 25 years », *Journal of Natural Products* 70(3): 461-477.

Pauly D. (2003), « Ecosystem impacts of the world's marine fisheries », *Global Change Newsletter*, 55): 21-23.

Pauly D., Christensen V., Dalsgaard J., Froese R. et Torres F. (1998), « Fishing down marine food webs », *Science*, 6, vol. 279, 5352, February, p. 860–863.

Pedroli G. B. M., van Elsen T., van Mansvelt J. D. (2007), « Values of rural landscapes in Europe: inspiration or by-product? », *NJAS Wageningen Journal of Life Sciences*, 54 (4), 431-447; http://library.wur.nl/ojs/index.php/njas/article/view/1343/962.

Piquet F. P., Blanc I., Corbiere-Nicollier T. et Erkman S. (2007), « L'empreinte écologique: un indicateur ambigu ("The Ecological Footprint: an Ambiguous Indicator")», *Futuribles, Revue d'analyses et de prospective*, n° 334, octobre, DOI: 10.1051/futur:20073345.

PNUD (2004), *Les Objectifs du Millénaire pour le développement (Millenium Objectives for Development)* www.undp.org/biodiversity/mdgandbiodiversity.html (accédé le 10/01/09).

PNUE (2005), *Perspectives mondiales de la diversité biologique, deuxième édition*, (*Worldwide Prospects Concerning Biological Diversity, 2nd Edition)* Secrétariat de la Convention sur la diversité biologique (Secretarial Office of the Convention on Biological Diversity), www.cbd.int/doc/gbo2/cbd-gbo2-fr.pdf (accédé le 10/01/09).

PNUE/Grid Arendal (2004), « Poverty-biodiversity mapping applications », *Discussion Paper* prepared for the IUCN World Congress, November, http://maps.grida.no/go/graphic/global-poverty-biodiversity-map (accédé le 03/02/09).

Rouault G., Candau J.-N., Lieutier F., Nageleisen L.-M., Martin J.-C. et Warzée N. (2006) « Effects of drought and heat on forest insect populations in relation to the 2003 drought in Western Europe »,. *Ann. For. Sci.,* 63 (2006) 613-624.

Sainteny G. (1993), *La Fiscalité des espaces naturels (Tax Policy applicable to Natural Areas)*, Éditions Victoires, Collection Environnement, 104 p.

Soussana J. F., Allard V., Pilegaard K., Ambus P., Ammann C., Campbell C., Ceschia E., Clifton-Brown J., Czobel S., Domingues R., Flechard C., Fuhrer J., Hensen A., Horvath L., Jones M., Kasper G., Martin C., Nagy Z., Neftel A., Raschi A., Baronti S., Rees R. M., Skiba U., Stefani P., Manca G., Sutton M., Tuba Z., Valentini R. (2007), « Full accounting of the greenhouse gas ($CO_2$, $N_2O$, $CH_4$) budget of nine European grassland sites », *Agriculture, Ecosystems and Environment*, 121 (1-2). 121-134, doi:10.1016/j.agee.2006.12.022.

Stern N. (2007), *The Economics of Climate Change: The Stern Review*, Cambridge University Press, 640 p. + annexes.

UICN France (2005), *La France et la biodiversité. Enjeux et responsabilité (France and Biodiversity. Stakes and Responsibility)*, brochure.

Wackernagel M. et Rees W. E. (1996), *Our Ecological Footprint: Reducing Human Impact on the Earth*, Gabriola Island, BC ; Philadelphia, PA: New Society Publishers.

Walker B. H., Carpenter S., Anderies A., Abel N., Cumming C., Janssen M., Lebel L., Norberg J., Peterson G. D., Pritchard R. (2002), « Resilience management in social-ecological systems: A working hypothesis for a participatory approach », *Conservation Ecology*, 6(1): 14, www.consecol.org/vol6/iss1/art14.

Weber J.-L. (2008), *Comptabilité des écosystèmes et de leurs services*, *in* session sur « Au-delà du PIB, à la recherche d'indicateurs synthétiques. Les relations entre environnement et développement durable », 12e colloque de l'Association de Comptabilité nationale (Compatibility of Ecosystems with their Services, in Session Dealing with "Beyond GNP, Looking for Synthetic Indicators. Relationships between Environment and Sustainable Development), 12th Symposium of the National Compatibility Association) Paris, 4-6 juin.

Weber J.-L., Uhel R., Spyropoulou R., Breton F., Arévalo J., Richard D., Haines-Young R., Potschin M., Kumar P., Martin B., Lomas P., Gomez E., Tomas P., Ezzine D., Nichersu J. et Marin E. (2008), *Draft report* on « Ecosystem Accounting for the Cost of Biodiversity Losses: Framework and Case Study for Coastal Mediterranean Wetlands », http://reports.eea.europa.eu/briefing_2008_2/en (accédé le 28/01/09).

# Chapter III

# The Legal Approach

**Two preliminary remarks are necessary in order to define what law can bring to thinking about the monetarisation of biodiversity.**

Firstly, thinking on this subject from a legal standpoint assumes that jurists are constantly following the statements and analyses of scientists and economists. But this attitude must not lead to our seeing law as a mere "toolbox" used to shape decisions or orientations defined elsewhere. If this warning is necessary, it's because this sort of perception and approach is frequent, especially in fields where the real or apparent technical nature of the scientific and/or economic analyses can give people the feeling that they are moving towards "objective" choices. This would be ignoring the fact that law necessarily develops its own discourse, even if it builds on the analyses of scientists and economists as well as the discourse of philosophers, sociologists and political scientists (among others).

**The so-called theory of autopoietic systems,** as it is used in social sciences and especially in law[1], takes this reality into account.

Very schematically, the upholders of this theory explain that there are various systems within society (political, religious, economic, legal, scientific, etc.) and that each of them is open to the others for the gathering of information (we then speak of "cognitive opening"), but they are then closed onto themselves for the processing of the information gathered with its own categories and concepts (which is referred to by the concept of "normative closure").

In this theory, the legal system is a system that is by nature very open to other systems, which "listens" to what is said by *all* of the other systems and which then translates this information into its own categories ("damage", "tort", "property", "entity", etc.). In this way the legal system will, for example, listen to what is said by the scientific and economic systems, and also hear what is said by ethics, morality and politics, etc. **In other words, the legal system is not and cannot be a mere "clerk" that receives economic and scientific analyses.** In this case there is no surprise that there can be a time lag between the scientific and/or economic analyses and their legal "translation." This lag is not a sign of the legal system's ignorance of these analyses but rather the simple result of the interweaving with other analyses and the taking into account of constraints from outside of the systems that produced them.

To illustrate this idea in a way that could seem provocative but which merely aims to clarify, it is possible to mention certain scientific and/or economic works on eugenics

[1] See Teubner (1988, 1994). See also the works of Niklas Luhmann, particularly Luhmann (1985).

or the death penalty: eugenics can have a scientific rationality, the death penalty may be more economically satisfactory than imprisonment, but both of them are contrary to human dignity and ethics, and law can, at a certain point, consider that it is socially more efficient to listen to what is said by the political or moral "systems" than what is said by economic or scientific systems.

**The second preliminary observation is both more technical and more simple:** it is a reminder that a judge can only rule on a request that is presented to him, which means that he cannot rule *infra* or *ultra petita*, and also that he rules based on the proof and demonstrations that are submitted to him. Therefore, the response of the judge can only be judged or evaluated in light of this remark: when an environmental protection association – FNE, for example – refuses to request reparation for ecological damage and prefers to only request reparation for moral damages, the judge can only respond to the question he is asked. The fact that he doesn't grant compensation for ecological damage does not necessarily mean that the jurisprudence is hostile to reparation for such damages. As the petitioner has chosen not to seek reparation for these damages, the judge cannot disregard that. Furthermore, when the judge evaluates damage which is by nature difficult to evaluate, he often does it based on the demonstration and the elements of proof provided by the parties, to which he may also add the opinions of experts.

Based on these two observations then, we can think about the contribution of law to the issue of "value" or "valuation", and likewise "monetarisation" of biodiversity, with the further specification that we will not reopen a discussion of the meaning of these expressions which will be taken in their commonly-understood meanings.

It was firstly within the framework of litigation for the reparation and – with a much more ambiguous meaning – in the field of prevention, that the issue of the value of biodiversity was first raised (III.1). But recent legal history has partly shifted the debate by giving great topicality to a concept that has been in our law for at least three decades: the concept of compensation. Compensation, by definition, invites comparisons of different situations and necessarily leads to an attempt to determine values of equivalence (III.2). There is then a progressive displacement of the stakes that must be underscored, and this displacement requires thinking on the legal disciplines involved (III.3).

## 1. The issue of the value of biodiversity in litigation for damages and in light of the ambiguity of the prevention texts

With regard to damage litigation, the purpose of the discussion that follows is not to present an exhaustive study of the jurisprudence concerning the reparation for damage caused to certain elements that make up biodiversity but to schematically describe the steps that have led to the current state of law. This presentation will demonstrate that certain responses were under construction and starting to take on a certain coherency and clarity when the new approach was imposed.

With regard to the prevention texts, a few lines will be sufficient to show that the legislators translated – often in monetary terms – the social reaction to the destruction of certain elements of biodiversity, without it being possible however to draw very strong conclusions about the "valuation" of these elements.

## 1.1. Reparation for ecological damage and its valuation by the courts

If we agree to schematise, stressing that the progression was not linear and that the various periods overlap and interconnect, **there were three steps in the evolution of litigation for reparation in the field under consideration.** The last one, in which the jurisprudence accepts the principle of reparation for "pure ecological damage" leads us to examine the modes of valuation of this damage.

### *a. The slow recognition of ecological damage by the jurisprudence*

**Initially, the judge only takes into account the harm to biodiversity by converting (in the sense of a currency conversion) the damage actually caused into economic damages in the strict sense***.* When asked to award compensation for damage caused by the discharge by an Italian company of "red mud" off the coast of Corsica, the judge to whom the case was referred, drawing on the estimations ordered on the approximate volume of water polluted and the average tonnage of fish caught by the fishermen of the sector involved, decided that the damage eligible for compensation corresponded to the loss of x tonnes of fish that could have been caught in this zone[1]. The same is true when, to award compensation for damages caused by an oil slick, a court of appeals converts this into "harm caused to the reputation of the tourist facilities of the coast"[2].

By using such conversions, which are obviously very limiting, the judge totally masks the damage caused to biodiversity and in fact accords it no value.

**In a second phase**, responding to the petitions of the plaintiffs, **the judge used, to translate the damage caused to elements of biodiversity, the very malleable notion of "moral damage".** In 1982, the Cour de cassation considered that the destruction of an osprey by hunters had caused a bird protection association "*direct personal moral damage in relation with the purpose and subject of its activities*"[3]. The moral damage claimed by the plaintiff stems from the harm caused to the elements of biodiversity that he has a statutory purpose of protecting (for example, the destruction of animals belonging to protected species – nocturnal birds of prey[4], chamois[5] – or pollution of the sea by hydrocarbons[6]), but it can also stem from the simple violation of a rule, even if it does not result in damage, if the aforesaid violation causes "direct or indirect harm to the collective interests that [the plaintiffs] have the purpose of defending"[7].

In these conditions, it is an understatement to say that the qualification of moral damage appears to be a sort of all-inclusive category which obviously does not take

---

[1] TGI Bastia, 4 July 1985, cited *in* «La réparation du dommage écologique marin à travers deux expériences judiciaires: Montedison and Amoco Cadiz», Gaz. Pal. July-August 1992, doctr., p. 582.
[2] Rennes Court of Appeals, April 18, 2006, n° 05/01063.
[3] Cass. 1re civ., 16 November 1982, Bull. civ. I, n° 331.
[4] CA Pau, 17 March 2005, n° 00/400632.
[5] CA Aix-en-Provence, 13 March 2006, *prec.*
[6] CA Rennes, 26 October 2006, n° 06/00757; CA Rennes, 18 April 2006, n° 05/01063; CA Rennes, 23 March 2006, n° 05/01913; T. corr. Brest, 8 March 2005, n° 04/000779.
[7] Wording of article L.141-2 of the environmental code.

into account the damage to biodiversity, the evaluation of which becomes pure guess work.

**In this regard at least, reparation for "pure ecological damage" – as it is called in the legal doctrine to distinguish it from the preceding qualifications – seems to be intellectually more satisfactory.**

Regardless of what was said by the press, which cited without verification the statements of certain lawyers or politicians, the acceptance of the principle of reparation for such damage by the courts does not date from the ruling made by the Paris Superior Court on 16 January 2008 in relation to the shipwreck of the *Erika*[1].

For several years now in France, trial judges have agreed to compensate such damage[2]. In 1988[3], the Criminal Court (for misdemeanours) of Brest ruled, to the benefit of a water quality protection association, that the destruction of fish in a river polluted by a drainage header caused direct and certain damage, particularly "in biological terms". Based on this analysis, the Court granted for this biological damage reparation that was independent of the "moral damage" also suffered by the association. In the same way, the Criminal Court (for misdemeanours) of Libourne in 2001[4] considered that due to the illegal dumping of a tank containing hydrocarbons, "the natural milieu [had] suffered unquestionable degradation", for which an association could request compensation. Still more explicitly, following work done without authorisation in the bed of a stream that led to the drying up of a river, the Court of Appeals of Bordeaux, in a ruling of 13 January 2006[5], compensated several associations for "damage suffered by the flora and invertebrates of the aquatic milieu" and "damage suffered by the aquatic milieu". Lastly, we should mention the ruling of the Narbonne Superior Court of 4 October 2007[6] – three months before the *Erika* decision – that compensated the damage suffered by a regional nature park due to the spilling of chemical products into the sea. It was this jurisprudence in the process of being formed[7] that was "mediatically" consecrated with the ruling made on 16 January 2008 by the Paris Superior Court in the case of the *Erika*.

In doing so, French judges apply solutions comparable in their principles to those that American judges, for example, have used on several occasions for damage caused to nature[8].

If we can consider that the principle of compensation for harm to certain elements of biodiversity, on the grounds of common law of civil liability, is accepted, it becomes

[1] On this decision, see Laurent Neyret, *Naufrage de l'*Erika*: vers un droit commun de la réparation des atteintes à l'environnement*, D. 2008, p. 2681.
[2] See the column of Laurent Neyret, *Compensation for damage to the environment by the judicial judge*, D. 2008, p.170. Although the *Erika* ruling had not yet been made, the author wrote: "the principle of compensation of damage to the environment by the judicial judge is now established". The illustrations that follow are drawn from the column of Laurent Neyret.
[3] Criminal court of Brest, 4 Nov. 1988, n° 2463/88.
[4] Criminal court of Libourne, 29 May 2001, n° 00/010957.
[5] CA of Bordeaux, 13 January 2006, n° 05/00567.
[6] D. 2007. AJ. 2731.
[7] "Being formed" only because thus far the Cour de cassation (supreme court of appeals) has not had the occasion to rule on such decisions.
[8] See in particular Brown (1981), Sands and Steward (1996) and Burlington (2004).

essential to consider the methods for evaluation of such damage, keeping in mind that what is observed for France will apply for the other States of the European Union.

### *b. Evaluation of ecological damage by the courts*

Firstly, we should stress that the "American" or contingent methods, based more or less directly on agreement to pay, have never been used in France or in Europe[1]. If European judges do not use these methods, it is certainly because plaintiffs do not ask them to evaluate the prejudice by these means. As for the question of why plaintiffs don't suggest these methods, we can imagine that they doubt – wrongly or rightly – that such evaluations, the scientific relevance of which is questioned by many economists, could convince a European judge.

In any case, the most frequently used methods can be grouped into three major categories.

**The first method, historically, was the composition valuation of the damage.** This is sometimes based on more or less official schedules, such as the one published in France by the ONCFS for game, which is regularly updated and often cited in court[2]. According to one author, the value used by the Office corresponds to the "cost of the reintroduction into nature of a number of individuals sufficient so that one of them can survive and replace the destroyed animal"[3]. In other cases, as in Spain[4] or in Hungary[5], there is a regulatory text that determines the schedule. It can also happen that the composition valuation is used without any explicit reference other than that made by the judge to equity. Very frequently, the reasoning of rulings is limited to stating "that the damage will be equitably compensated by allocation of the sum of…". Even if the justification is not this succinct, it was in this fashion, without mention of any criteria for evaluation, but simply referring to "the scope of the pollution" that affected "birds belonging to many different species" thereby causing "a real ornithological disaster", that the Paris Criminal Court evaluated "the damage resulting from harm to the environment" cited by the Bird Protection League (LPO) in the case of the *Erika*. We know that the damages for the estimated loss of 60,000 birds were valuated at 300,000 euros. The application of a schedule inspired by that of the ONCFS (30 euros for a pigeon – the least "expensive" of the birds of the list –, but 800 euros for a rock partridge, for example) would have given quite a different result! It is likely that the same would have been true with the application of contingent evaluation methods which, whatever their limitations, cannot be considered to be less scientific than a guesswork evaluation!

**The second method, which in many ways seems more satisfactory, is based on the cost of repair or restoration *in situ*.** Ruling on the reparations due to a National Park for the damage resulting from the picking of *génépi* (artemisia), the judge took into consideration the cost for the operation of finding on the deteriorated site approximately the same number of sprigs as if the destruction had not taken place. To

---

[1] See note below.
[2] Schedule of the values of various species of game intended to be used as a basis for requests for damages before the courts, updated by decision n° 07/01 of the Board of Directors of 12 April 2007. See www.oncfs.gouv.fr/events/droit_jurisprudence/Bareme_valeur_gibier_2007.pdf.
[3] Neyret L., 2008, *Naufrage de l'*Erika, as cited above.
[4] Decree of January 22, 1986.
[5] Application regulation of 15 March 1982 of Decree -Law n° 41-1982.

do this, there was a calculation of the cost of the work needed on site to gather *génépi* seeds with the same genetic characteristics, to give these seeds to an INRA laboratory in charge of cultivating them, and lastly to bring the plants to the site and to monitor them, taking into account the inevitable losses at each step in the process. By way of illustration, in 2004 such a method would have led to evaluating each sprig of *génépi*[1] illegally picked at 3.05 euros.

**The third method involves valuating the damage by reference to a budget spent as a total loss to manage the natural goods that were destroyed and for which the plaintiff was responsible.** In order to evaluate the damage caused by the destruction of chamois, the court could, for example, isolate in the accounting of a national park the budget devoted to the management and protection of hoofed animals, and then apply to these sums a ratio taking into account the number of animals lost (which includes the animals destroyed and their descendants over an arbitrarily determined period) with respect to the number of animals living in the area managed by the park[2].

This method is similar to that used by the Paris Superior Court in the *Erika* case to evaluate "the damages resulting from harm to the environment" suffered by the Morbihan Department. The judges, citing the demonstration of the department, combined a fiscal criterion, taken from the amount of the departmental tax on sensitive nature areas for the year 2000 (2,300,000 euros), a spatial criterion taking into account the fact that 662 hectares of sensitive nature areas out of the 3,000 hectares belonging to the department were damaged by pollution, and a temporal criterion, considering the two years over which the effects of the pollution continued. With this evaluation method, the judges granted for this damage the sum of 1,015,066.60 euros [(2,300,000: 3,000) x 662 x 2].

As already underscored, **these responses are far from having been stabilised and, with regard to taking into account the harm directly caused to elements of biodiversity, they have never been confirmed by the Cour de cassation.** Nevertheless, this "progress" could have been considered as serious and well "established" in the European legal landscape when there came the thunderclap of directive 2004/35/CE concerning environmental responsibility. These initial responses could be fundamentally challenged[3] by the emergence of a new approach and by the (re)discovery of the principle of compensation.

Before discussing this possible shift – or in any case, this new data –, we should point out in a few words that criminal law, with its own imperatives and the objectives, has frequently given a costed monetary translation of the destruction of certain elements of biodiversity.

---

[1] There are several sprigs on one plant.
[2] See, for example, Aix-en-Provence CA, March 13, 2006, n° 428/M/2006, unpublished.
[3] Even if the debate is full of doctrine and although many authors consider that it should not go this way. See in particular Camproux-Duffrène (to be published). See also Guihal and Nesi (2007), p. 230 and following.

## 1.2. Punishment of harm to biodiversity: a monetary translation of the social reaction

As mentioned by President Bruno Cotte in the preface to the third edition of the book of Dominique Guihal (2008), **penal sanctions "reflect a social reprobation that is not linked to measures of coercion or of administrative or civil reparation"**. For this reason, it can be tempting to seek the penalties that are provided for by law to punish harm to certain elements that make up biodiversity. However, it seems difficult to derive from this any lessons that go beyond mere observations.

Article L. 411-1 of the environmental code establishes the prohibitions intended to guarantee "the conservation of non-domesticated animal species and non-cultivated plant species" for which the regulatory authority, by the terms of article L. 411-2 of the same code, sets the list and for which it determines the extent of the protection[1]. As for the penalties, they are mostly presented in articles L. 415-3 and R. 415-1 of the environmental code. The first of these texts punishes with "six months imprisonment and a fine of 9,000 euros, in violation of the prohibitions provided for by the provisions of article L. 411-1 and by the regulations established in application of article L. 411-2, a) the harming of the conservation of non-domesticated animal species, (…); b) harming the conservation of non-cultivated plant species (…)". Since law n° 2006-436 of 14 April 2006, the fine is doubled when these infractions are committed within a National Park or nature reserve. Article R. 415-1 punishes with penalties provided for 4th class offences (750 euros at most): "1° Intentionally disturbing non-domesticated animal species protected under L. 411-1; 2° Introducing into the natural milieu, by negligence or by carelessness, any specimen of a species, animal or vegetable, mentioned in article L. 411-3 (…)".

What can we conclude from this? Evidently that lawmakers have given a monetary translation to certain types of harm to elements of biodiversity that the regulatory authority considers to be worthy of protection. But can the level chosen be considered as giving an indication of the value of the biodiversity thus protected? We have doubts about this for at least two reasons. First, the level of the penalties is determined above all by the nature of the infraction – misdemeanour or offence – and falls within a scale that has little to do with the seriousness of the damage caused. Other than in exceptional cases, criminal law does not aim to repair the consequences of the harm, but rather to suppress the social disorder caused by the infraction. With this approach, the same penalties are applied to the destruction of an animal belonging to an endangered species, its transport – even if the animal is already dead – and the destruction, alteration or degradation of the particular milieus of certain animal or plant species. Furthermore, the maximum threshold of the penalties provided for in the texts is only exceptionally reached in the field, with the judges having to take into account the circumstances of the infraction, the personal situation of the accused party, all elements that obviously have nothing to do with the extent of the damage caused.

That is why it is important to return to the heart of the matter by analysing the shift that is occurring between reparation and compensation.

[1] See Guihal D. (2008), p. 481 and following, especially n° 41 202 to 41 204.

## 2. From reparation to compensation

A radical change occurred with the adopting of directive 2004/35/EC concerning environmental responsibility and law n° 2008-757 which transposed it into French law. At the same time there was a rediscovery of older texts which had been largely neglected which it would be good to see "reactivated". This double evolution invites us to consider the emergence of the concept of compensation and its consequences.

### 2.1. Directive 2004/35/CE concerning environmental responsibility and transposition law n° 2008-757 of 1 August 2008

We cannot analyse these texts in detail here. We will merely seek the elements of their provisions that could nourish the thinking of this report. Prior to their examination, it is important to stress that, despite what the title could lead people to think, **these texts do not establish a "responsibility" in the usual sense of the world, but rather an administrative policy for the purpose of preventing and/or repairing certain damage to the environment.**

With this point made, **two essential points deserve to be highlighted** for thinking about the valuation of biodiversity.

**Firstly, the directive and the transposition law formally consecrate the principle of the necessary reparation of *certain* damage to the environment that the French text, faithfully following the directive, lists and defines in what has become article L.161-1, I of the environmental code.** This is the main contribution of these texts: the obligation to repair the "pure ecological damage" is now inscribed in law, at least for the categories of damage that enter into its projections. The various categories of damage involved are damage to the soil (in the case of and only in the case of risk of serious harm to human health), serious harm to the ecological, chemical or quantitative state of water and, lastly, harm seriously affecting the state of conservation of populations of species of wild fauna and flora protected by law. In all cases, verification of the "seriousness" of the harm or risk of harm is a prerequisite for the application of the new system. It is lastly worth pointing out that the same article L.161-1 introduces into French law the notion of "ecological services" and submits the losses affecting the aforesaid services to the new system.

**Secondly, the directive and the transposition law formally exclude reparation by a monetary equivalent** and define two systems for reparation to which a third one was added.

The primacy of repair under the auspices and control of the administration is first established. That's what the law means by the term "primary reparation".

When restoration is impossible, compensation by an equivalent in kind (possibly at another site) must be chosen by the competent authority. This is what the law, again following the directive, rather curiously calls "complementary reparation", probably considering that, in most cases, this form of reparation will not take the place of a totally impossible primary reparation, but will supplement a partial restoration, but out of the area.

Lastly, in addition to these two forms of reparation there is what the law designates under the term of "compensatory reparation" and which involves compensation by an in-kind equivalent of the ecological services lost between the time of the damage and the time when the primary and/or complementary reparations took their effects and allowed for a return to "normal".

**Compensation in kind, which implies evaluations (not necessarily monetary of course[1]) and then the determination of equivalence values, has thus become one of the major tools for reparation of damage to the environment.**

## 2.2. The rediscovery of compensation in national and EU texts

At the same time, we seem to have rediscovered that **various texts, of EU and/or domestic origin, provide that a development, an activity, a project, a plan or a programme, that could harm the environment and particularly biodiversity, can only be authorised by the government if the petitioner does everything that is necessary to avoid the impact, reduce the negative impacts that he cannot avoid and, lastly, "if possible" the texts add, compensate for the residual negative impact**. This three-fold approach – avoiding, reducing and compensating if possible – has been present since 1976 in French law (today articles L.122-1, and L.122-3 of the environmental code) and was again consecrated by EU law in directives n° 85/337, called the "projects" directive, and n° 2001/42, called the "plans and programmes" directive. A partial translation of these directives has since then been integrated in the environmental code in article L.122-6. The obligation of compensation for the residual negative impacts that is present in the texts has thus far only very exceptionally been applied and often at the initiative of field agents, who are frequently worried about implementing compensation operations with no guidelines other than their own common sense and their knowledge of the sites involved. Although they don't always dare to express it, these field agents really want a legal framework for such operations[2].

**In light of what has just been explained, it can be said that compensation has become a central concept (we wouldn't say unique) of the legal system in cases of harm to elements of biodiversity, whether this harm has already occurred, or if the authority in charge of forbidding, authorising and imposing requirements anticipates it.**

How can the law interpret this paradigm shift?

## 2.3. A paradigm shift

There is an initial vital question: is it legitimate to compare provisions involving damage that has not yet occurred and that the government can theoretically prevent by refusing the petitioner the authorisation that he is requesting, and those for reparation of damage that has already occurred? We are tempted to answer yes. But

---

[1] We will come back to this point in III.2.3.

[2] For this point we refer to the presentation that was made of some of these experiences at a colloquium held at the Cour de cassation on 24 May 2007 on reparations for damage to the environment: www.courdecassation.fr/formation_br_4/2007_2254/assurances_responsabilites_9490.html; for a summary presentation of this work, see Martin (2009).

comparing does not mean equating. The fundamental difference between these two situations is that a decision to forbid is still possible in the first case – and must be applied if the common interest justifies it – while, by hypothesis, it is not in the second case. However, beyond the fact that the reality in the field probably involves more nuances, this comparison is necessary at least with regard to the concept and the implementation of compensation, when it must occur. It must in the system established by the directive and the law on environmental responsibility for damage on the point of occurring or which has already occurred; but logically it also should after examination of the request and verification that the petitioner has made his best efforts to avoid the negative impacts and to reduce those that cannot be avoided. The competent authority will decide that the operation deserves to be authorised for reasons of general interest – the appraisal of which could be under the control of the judge –, on the condition that the petitioner is able to compensate for the residual negative impacts.

In other words, it would be paradoxical to note that the reference to "compensation" is present in the legislative discourse for both hypotheses and to consider, at the same time, that no comparison should be made.

**Within the framework of this renewed approach, lawyers encounter a concept that they know well and that they use often: fungibility**. In obligation law, compensation is only possible among things that are fungible. By definition, the elements of biodiversity, other than in exceptional cases, are not fungible amongst themselves. In this case, the only way to apply compensation between non-fungible goods is to find a common measure, a common reference value that allows for comparison of things that are incomparable. This common reference can be built from very diverse indicators, the ultimate aim of which must be to allow for comparison and, when possible, compensation. That is why, in obligation law, fungibility is most often provided solely by the most fungible thing there is: money. Is this a problem? What can law bring to thinking on the monetarization of biodiversity?

**Contrary to the hasty conclusions we might draw, jurists do not necessary have qualms about converting everything into monetary values**: they do it every day when they agree to grant damages in reparation of harm to honour, to physical integrity, to image or to private life, etc. They also do this when they accept the subscribing of life insurance policies or insurance against feared illnesses. In this respect, the fact that reference values are assigned to elements of biodiversity to clarify public decisions or to guarantee better application of the jurisprudence of the "cost/advantage" balance initiated by the State Council in 1971 with the Ville Nouvelle Est ruling[1], should not come as a blow to principles. The doctrine often indicates that this jurisprudence, which involves the adjudicator comparing the collective "costs" of a project with its "advantages", to evaluate the legality of a declaration of public utility, did not live up to the hopes placed in it. More specifically, we see that judges quite often underestimate the "costs" of the project, particularly the ecological costs, when, for example, they need to do an appraisal of a major infrastructure project of national interest[2]. Contrary examples exist[3], but the reality is indeed what is described. It is

---

[1] C. E. 28 May 1971, Rec. page 409, concl. Braibant.
[2] See for example Chrestia (1997), p. 545; Hostiou (2006), p. 604.
[3] C. E. 20 October 1972, *Société civile Sainte Marie de l'Assomption, Rec. page 657, Conclusions Morisot;* C. E. 25 July 1975, *Syndicat des marins pêcheurs de la rade de Brest,* RJE, 1976, p. 63*;* C. E. 28 March 1997, *Autoroute A 400 Annemasse-Thonon,* Rec. 210 (but this cancellation was

possible to add that in rather rare cases when the judge takes into account the environmental "cost" of a project, it is not the harm to biodiversity in the strict sense that swings his decision, but rather the landscape appearance and the protection of the sites. In this sense, giving biodiversity a monetary valuation could contribute to making it more visible by giving it the place that it deserves in the evaluation done by the judge.

**But can we accept going further and using the reference value for the purposes of exchange and compensation? That is another issue entirely.** The fact that the judge grants an indemnity because a medical error deprives the victim of a kidney does not mean that the kidney is merchandise that can be traded. In other words, the law can have reservations about accepting that certain "things" be "legal goods" that take part in legal commerce and trade. For this to be more than just words, it is necessary that the law expressly say so and that an authority (which could be the judge, the State, or both of them), make sure that the principle of non-commerciality is respected.

The position of the legal system will sometimes be more nuanced: the prohibition on trade could include the possibility of exceptions when decided by an authority and within the limits provided for by law. An administrative authorisation is therefore not in the legal commerce, but taxi licences do involve business. There is a market for authorisations for the discharge of greenhouse gases converted into quotas.

In these hypotheses – when the goods are by principle or exceptionally in the legal commerce and a market can exist for them – a new consideration must come into play: some markets involve goods which, in addition to their market value, also have values from outside of the market (radio waves are also vectors of freedom of expression, for example). **If the law accepts that trading of these goods can take place – which is an absolute prerequisite -, there must be thinking on issues of the regulation of the trade.** The questions asked are then questions of procedure, in the strong meaning of this term, i.e. issues of democracy.

This is typical of what is happening with biodiversity, taking us from compensation law into regulation law.

## 3. From compensation to regulation

A rapid and very schematic presentation of the projects underway will precede the necessary thinking on the functioning and regulation of the system.

### 3.1. Projects underway

In 2008, the Caisse des dépôts established a 100% subsidiary (CDC Biodiversity) with a capital of 15 M€, the purpose of which is to support operators in this field. Among other accompaniment measures, the company decided to "produce" biodiversity,

---

absolutely not based on the taking into account of the environmental "costs", as the judge had even expressly excluded them from his evaluation: see Chrestia (1997), *loc. cit.;* C. E. 10 July 2006, THT Boutre-Carros, RFDA, 2006, p. 990, note M.-F. Delhoste.

particularly in order to offer "units of biodiversity" to operators who would have to compensate the harm that their activity caused or could cause. According to the people in charge, this part of the activity of the new subsidiary currently represents about 10% of the total revenues.

The rough outline of the scheme is the following.

The CDC acquires areas that it manages or has managed (by operators approved by the State) in a spirit of conservation and protection. It thus proposes to "manufacture" biodiversity.

These spaces and this management are then "converted" (under the responsibility of and with the approval of the State) into spatial-temporal units of account called "units of biodiversity" (the "ecological" management for 30 years of 100 hectares corresponding to the habitat of a given species will represent, for example, 50 units of biodiversity).

When a developer or promoter of an activity plans to carry out a project, the State, after verifying that the developer/promoter has done everything to (i) avoid the impacts, (ii) reduce the impacts that could not be avoided, will note the residual negative impacts. These residual impacts will be converted into units of biodiversity and the developer/promoter can acquire from the biodiversity manufacturer as many units as he destroys.

An identical system could be used for reparation of harm to the environment covered by the law of 1 August 2008: as a complementary and/or compensatory reparation, the person responsible for the harm could consider "acquiring" a number of units of biodiversity equivalent to those that he destroyed and that could not be covered by a primary reparation.

## 3.2. Plea in favour of the regulation of compensation and trading mechanisms

The experience described above, largely inspired by mechanisms already implemented, particularly in the United States[1], obviously gives rise to a whole series of questions of principle. What will be the legal nature of these units of biodiversity? Is it possible to discuss this point with reference to the administrative authorisations that grant greenhouse gas quotas? How and by whom are values of equivalence established between "things" which, by nature, are not fungible? If these units of biodiversity correspond to a commitment of "ecological" management that is limited in time, isn't there the risk of the same unit being used to compensate several successive damage incidents[2]?

All of these issues are legitimate and merit consideration that should involve legal theory, property law and guidelines of environmental law. All of them also involve the issue of the regulation of such a market. It is important to consider them from this latter angle within the framework of this report, because any new market calls for the establishment of regulation mechanisms and entities.

---

[1] See, for example, Hernandez (2004).
[2] On all of these issues, see Camproux-Duffrene (2008), p. 87.

If we accept a schematic view[1], the expression "new market" can refer to two situations.

There can be a new market through the opening to competition of a sector that had been a public monopoly (television, telecommunications, to cite only the best known cases). The new market can also arise because the goods, which used to be outside of the market, newly enter a market. This is the case for greenhouse gas quotas and future units of biodiversity.

The implementation of mechanisms and regulation bodies is all the more necessary when there is a need to strike a balance between free competition to be established and other heterogeneous market principles. For television, for example, it is necessary to find a good equilibrium between free competition and freedom of expression, artistic creation, etc.[2]

The market for units of biodiversity corresponds very exactly to this scheme. The idea is, by creating deeds that represent it, to convert biodiversity into instruments that can be used in trade and compensation mechanisms. These new goods, the objects in the trade, are in themselves particularly difficult to define. All specialists agree that biodiversity can no longer be seen as a simple tally of genes or species; it is the interactions between these elements, the services that they provide[3], which are essential. The first difficulty is (will be) thus "making" the goods in question.

Beyond that, these new goods obviously involve values from outside the market which are difficult to determine. To facilitate our analysis, we will put them under the heading of "sustainability", in order to express the idea that the goods represented by the deed are not – in the strict sense – infinitely reproducible and also that they are part of the equilibria necessary for life.

The schematic analysis presented above shows risks of confusion between the authority prerogatives of the State[4], the determination of trade or compensation values[5] and the surveillance of the market thus created, particularly to avoid all malfunctions, such as collusion or dominant positions, which are certain to arise.

This functional confusion could lead to a series of undesirable effects, above all that the State authority prerogatives become subject to a commercial rationale – the project is authorised because the damage can be compensated through the purchase

---

[1] For further nuances, see the fundamental article of M.-A. Frison-Roche, "Regulation law", *D.* 2001, p. 610 & s.

[2] "A sector that justifies the construction of a set of rules and a regulation authority is in the space between competition law and public law, in a dynamic and intersecting conception of both… The sector is both open to competition and not abandoned to competition," writes M.-A. Frison-Roche (*ibid.*). See also, from the same author, "Definition of economic regulation law", *D.* 2004, p.126 and following.

[3] Directive n° 2004/35/CE concerning environmental responsibility and the law of 1 August 2008 which transposes it, introduced this concept into domestic law.

[4] Forbidding a project, authorizing it, applying the administrative policies to which it will be subjected, accepting or refusing a programme of reparations presented by the party responsible for the harm.

[5] Which implies the establishment of "equivalences", or even conditions for the establishment of these "equivalences".

of units of biodiversity – and still worse, the risk of having the commercial rationale completely dominate the exogenous non-commercial values.

**In other words, it seems essential that the authority which will operate the market and will establish the values of equivalence be distinct from that which will exercise the authority prerogatives and determine what is in the sphere of the exchange.**

**As is frequent, the seeds of such a regulation authority are present in a whole series of current initiatives, even if no project presents all of the necessary guarantees.** Analysis of the practices of these various institutions in gestation will allow us to determine whether they intend to play the role of regulator.

Can this role of regulator be exercised by the Scientific Council that the Caisse des dépôts et consignations decided to establish? There are reasons to have doubts about this: the main quality of a regulator is that he is outside of the regulated sector, while representing the contradictory interests that are expressed within it. It is chiefly through this exteriority and the acceptance of the decisions that he makes that he establishes his legitimacy[1].

The establishment of a "compensation observatory" which is now said to be scheduled could meet the condition of exteriority, but its objective would *theoretically* – as its name indicates – be limited to observation, which cannot be the only mission of a regulation body.

European directives 85/337 ("projects") and 2001/42 ("plans and programmes") went part of the way towards the constitution of a real public regulation system: they provide for a **"competent authority for the environment"** that issues its opinions, which are made public, through a process of elaboration of the decision on the evaluation of the environmental impacts presented by the petitioner of the operation, and in particular **on the measures considered to avoid, attenuate, or compensate for these impacts**. These provisions thus amount to having a qualified authority consider the measures considered, if any, in terms of compensation.

By virtue of the principle of subsidiarity, these directives leave to the Member States the choice of the modalities for the implementation of these provisions. Their transposition into French law, which is now being completed, assigns the "environmental authority" function either to the prefects for operations of local scope, or to structures linked to the ministry in charge of the environment for operations on a national level. Furthermore, the function of this "authority" is currently limited to the issuing of the rulings provided for in the directives.

**Over time, the implementation of an efficient regulation system would suppose that it be truly independent and that its competences explicitly extend to validation of the compensation operations, which is only implicit at present.** In this regard, if, in order to facilitate the functioning of this market, recourse to monetary valuations on the model of the "shadow" values already existing for human life or the passing of time can be considered, the central issue is determining the procedures according to which these values will be elaborated and which contradictory interests

[1] See the work of M.-A. Frison-Roche, prec.

will be taken into account. It seems clear that technical legitimacy is not sufficient and that such values cannot be socially accepted unless their elaboration benefits from democratic legitimacy. The environmental regulation authority could be the instrument for this.

These conditions are essential for the emergence of a real regulation authority. For such an authority to be useful and effective, it must be independent of the executive and the sector that it regulates, while being capable of gathering all information from this sector and taking into account the contradictory interests that are expressed within it.

If, taking this thinking further, we consider the composition of such an authority, several proposals can be made. Magistrates of both orders should clearly be part of it; alongside them there should be representatives of the State, non-governmental environmental protection organisations, developers and promoters, and also qualified personalities from the scientific communities involved, without forgetting the human and social sciences.

Other difficulties merit further thinking: what should be the prerogatives of the authority? Beyond the “influence” that it could have through the publication of reports or opinions, should it have regulatory prerogatives? Could we even consider giving it jurisdictional prerogatives? In this latter hypothesis, what recourse should be considered? An organisation providing regulation functions is only useful if it provides stable references which can be audited and which are homogeneous, based on all of the assembled knowledge for the field that it examines. This type of organisation is constantly exposed to the risk of “capture” by certain actors of the sector and one of the most effective defences against this risk lies in publicity and auditing of the opinions that it issues, based on the principle of objectivity, in order to facilitate the controversy that will be useful for the final decision.

Thinking on these various issues should not be postponed for too long.

## Bibliographic References

Brown E.D. (1981), « Making the polluter pay for oil pollution damage to the environment: a note on the Zoe Colocotroni case », *Lloyd's Marine and Commercial Review Quarterly*.

Burlington L. B. (2004), « Valuing natural resource damages: a transatlantic lesson », *Environmental Law Review,* 6 (2).

Camproux-Duffrène M.-P. (à paraître) (to be issued), *Les modalités de réparation du dommage; les apports de la responsabilité environnementale,* actes du colloque du Mans (Methods of Damage Compensation, Proceedings of the Le Mans Symposium; Contribution of Environmental Responsibility (2008, November 27th-28th), Paris, Dalloz.

Camproux-Duffrene M.-P. (2008), « Le marché d'unités de biodiversité, questions de principes, *Revue Juridique de l'Environnement*, numéro spécial.("The Market of Biodiversity Units, a Matter of Principles, *Environment Legal Aspects Magazine*, Special Issue)

Chrestia P. (1997), *Application de la théorie du bilan en matière de travaux autoroutiers,* AJDA.( Application of Results Theory in the Field of Motorways Works)

Guihal D. (2008), *Droit répressif de l'environnement*,(Repressive Legal Rules applicable to Environmental Matters) Paris, Economica.

Guihal D. et Nesi F. (2007), « L'articulation du nouveau dispositif de responsabilité environnementale avec le droit commun »("How the New Environmental Responsibility Scheme is Linked with Common Rules") *Droit de l'environnement*, (Environment Law) n° 151, p. 230 et s.

Hernandez S. (2004), « Compensations pour la diversité biologique, une opportunité économique et financière »(Compensations for Biological Diversity, an Economical and Financial Opportunity) , Revue *Échanges (Echanges* Magazine*)*, décembre, n° 238.

Hostiou R. (2006), *La théorie du bilan à l'épreuve des atteintes à l'environnement et à la propriété privée,* AJDA. (The Results Theory Facing Damages Inflicted to Environment and to Private Property)

Martin G. J. (2009, à paraître (to be issued), « La réparation des atteintes à l'environnement » , *in La réparation des préjudices*,("Compensating for Damages Inflicted to Environment" in *Prejudice compensation*) Paris, Dalloz.

Luhmann N. (1985), « The autopoiesis of social system »*,* Florence, EUI, *Colloquium Papers* (Doc. IUE 328/85 – Col. 81).

Sands P. et Steward R. B. (1996), « Valuation of environmental damage – US and international law approaches », *Review of European Community and International Environmental Law,* 5 (4).

Teubner G. (1994), *Droit et réflexivité: l'auto-référence en droit et dans l'organisation*, Paris, LGDJ.

Teubner G. (1988), *Autopoietic Law: a New Approach to Law and Society,* Berlin/New York, Walter de Gruyter.

# Chapter IV

# State of knowledge: biological concepts and indicators

## 1. The concepts of biodiversity

The term biodiversity appeared relatively recently, in 1985, in works on the diversity of living things[1]. For some, it is just a simple fortuitous semantic innovation that has certainly succeeded, but that does not introduce any new concept. Others (Aubertin, 2000) have emphasised that this new term marked the removal of these problems from the strict field of interest of biologists and protectors of nature and their entry into the political field, with the result that the questions raised become loaded with both promises and threats, and strongly involve man.

This term has "crystallised" several long developments in the perception of nature and introduced several important concepts, which must be taken into account in indicators that aim to describe and measure biodiversity. A more detailed presentation of these different concepts is proposed in papers by Chevassus-au-Louis (2007, 2008a and 2008b).

Among numerous possible definitions, the working group selected that proposed by the Rio Convention (1992): "variability of living organisms of all origins, including, among others, the land, marine and other aquatic ecosystems and the ecological complexes of which they are part: this includes diversity within species and between species as well as that of ecosystems".

### 1.1. The premises: making an inventory of Noah's Ark[2]

The premises guiding naturalists' thinking go back to 1758, a date considered as founding modern systematics. This is in fact the date of the publication by Linnaeus of his tenth edition of *Systema Naturae*, which describes species by the binominal nomenclature (genus then species name), a nomenclature principle since universally accepted. About 10,000 species are described in it, 6,000 plant species (essentially land plants) and 4,400 animal species, of which about a third are vertebrates. At that time, the fixist idea, according to which species were the work of the Creator and had remained immutable since their creation was not contested, even if the fact that some of them could have disappeared in catastrophic events was recognised. Making an

---

[1] The word is the contraction of the expression "*biological diversity*", itself invented by Thomas Lovejoy in 1980. It was proposed by Walter G. Rosen in 1985 during the preparation of the "National Forum on Biological Diversity: and appeared for the first time in a publication in 1988, when the American entomologist E. O. Wilson made it the title of the report of that forum.

[2] This reference can be taken literally: when Louis XIV sent the botanist Joseph Pitton de Tournefort on an expedition to the Levant in 1700, the latter did not fail to climb Mount Ararat, in search of any remains of Noah's ark.

inventory of living things, ordering species, was thus rediscovering the Creator's plan, as explicitly explained by Linnaeus: "Deus creavit, Linnaeus disposuit".

## 1.2. The new frontier: the inaccessible great inventory

Although it was little by little freed from this metaphysical dimension, this great inventory project motivated the efforts of naturalists throughout the XIX[th] and XX[th] century, arriving today at the figure of 1.7 million species described. However, a major change of perception took place in the 1960s, linked to new methods allowing more systematic exploration of certain ecosystems (tropical forests, the deep ocean) and an estimate of the number of species still unknown. Whereas the great inventory had been considered largely completed, which had also lead to a loss of interest by the scientific community in these activities, this work in fact revealed that known species only represented a small part of the total of all species and that, in addition, this visible part was no doubt not representative of what remained to be discovered. In fact, if groups like vertebrates and terrestrial plants could be considered as well inventoried, most invertebrate groups, algae or fungi were doubtless essentially still to be explored. In other words, the perception of biodiversity was based on relatively large, easily observable species, which were more the exception than the rule among living things[1].

This inaccessible[2] character of the great inventory increased still more when modern methods of characterisation, based on molecular biology, could be applied to micro-organisms, in particular bacteria. By analogy with astronomy[3], we can in fact talk of the "dark matter" of biodiversity, because it is now evident that these microorganisms represent, on the planet, as much in terms of biomass as diversity and functional role, the essentials of biodiversity. As the palaeontologist Stephen Jay Gould[4] wrote, "The paradigm for the success of life has always been bacteria".

**This therefore leads to the formulation of a first challenge: that of handling the unknown, meaning describing biodiversity and its changes from a perception that is not only very partial, but also very biased in its reality.**

## 1.3. The evolving vision and its consequences

A muddled intuition of the Encyclopaedists[5], the fact that current species were not immutable but represented a sort of "freeze frame", meaning the expression, or rather one of the expressions, at a given moment of the dynamics of life was a major gain of the naturalists of the XIX[th] and XX[th] siècle, in the first rank of which the figure Charles Darwin stands out. Limiting ourselves to the question of biodiversity, this 'Copernican

---

[1] If we take the example of "shellfish", dear to collectors, we now know that the great majority of these species have shells that do not exceed a few millimetres in length.
[2] Taking the minimum hypothesis of 10 million species and considering that currently 10,000 species are being described per year, the end of the great inventory will occur in… 2830.
[3] It is currently estimated that the greater part of the matter in the universe is not observed because it does not emit detectable radiation, hence the term dark matter.
[4] Gould S. J. (1997), *L'Éventail du vivant (~ The Spectrum of living things)*, Éditions du Seuil, Paris, p. 210.
[5] *Le Rêve de d'Alembert (~ Alembert's Dream)* by Diderot (1769) is often cited as one of the first expressions of this evolutionary thinking.

revolution" introduced two concepts that still sometimes seem to be insufficiently considered today.

**The first is that of the crucial importance of diversity within species, most often described, in a restrictive manner, as genetic diversity**[1]. In fact, in the fixist concept – and Linnaeus expressed himself very clearly on this point – diversity within species was not unrecognised, because naturalists had the multiplicity of animal races or variety of cultivated vegetables before their eyes, but this was considered as an epiphenomenon, often linked to human activities, which introduced limited variations around a "type", the work of the Creator (from which comes the idea of the "typological concept" of the species). The inspired intuition of Charles Darwin, linking these two phenomena and postulating that the variation between species was constructed over the long term from variation within these species, radically changed the status of genetic diversity, which then appeared not as an artefact but as the heart of the capacity of species to adapt and evolve. From which arises the need to describe this level of organisation of diversity and to monitor its changes.

**As a second consequence, this vision of change leads to an emphasis on the limits of a species inventory as a measure of biodiversity.** Due to the changing dynamic, certain species could have diverged very recently and thus be very similar, whereas other could have a very marked, old evolutionary divergence. From this, rapidly evolving groups, like insects, could present a very large number of species without us being able to say that their biodiversity is more marked than that of groups like mammals that evolve more slowly. Added to this is the fact that the tendency of systematicians to create species and the criteria that they use to do this can vary strongly from one group to another and that it sometimes needs a long time to verify that these species really meet the criteria for a "good" species, in the biological sense of the term[2], or that they have not already been previously described under another name. As a result two species drawn at random will not measure the same "quantity" of diversity, in other words, measuring the biodiversity with a rule "graduated" in species will give very different results depending on the part of the rule used.

We may think that in the future, methods from genomics will measure evolutionary divergences between species more directly and unify the ideas of genetic diversity and species diversity into one concept. But, for the moment, we must continue to use inventories of species identified by easily observable characteristics to describe species diversity, whilst being aware of these limitations.

## 1.4. The contributions of functional ecology

In a way fairly independent from the emergence of evolutionary thinking, the development of ecology and in particular of functional ecology[3], has provided a better understanding of the functioning of ecosystems and the role played in them by

---

[1] In fact this term is too restrictive: there are other aspects, like the diversity of behaviours between animal populations, which are transmitted by learning and can play an important role in the ability of populations to adapt.
[2] We refer readers to specialised works for a discussion of this biological concept of the species.
[3] Branch of ecology that studies ecosystems from the viewpoint of major exchanges of matter and energy between the living things that populate them.

different living organisms, most often considered in terms of homogenous species[1]. As before, the working group has limited itself to identifying new concepts from such work that should now be taken into account in the concept of biodiversity.

**The first concept, which now seems obvious, is to consider the diversity of ecosystems and the distribution of living things on the planet as a key dimension of biodiversity.** Whereas Linnaeus saw this distribution as the immutable result of divine providence, ecology aims to understand the determining factors, in particular environmental factors. The famous map of the altitude separation of vegetation on the flanks of the Chimborazo volcano (figure IV-1), from the ascent by Von Humboldt and Bonpland in 1802 of what was considered at the time as the highest mountain in the world, is often held out as the founding act of this determinist vision of the distribution of biodiversity. More recently, these strictly deterministic approaches have been modulated by taking the historic and contingent dimension of ecosystem populations into account (see for example Hubbel, 2001): to paraphrase Jacques Monod, ecosystems will now appear as the products of chance and necessity.

**The second concept is that of the importance of functional interactions of all orders between species, linked to multiple exchanges**: food exchanges, which ensure the circulation of energy within the ecosystem, but also exchanges of physical or chemical signals that will confer specific emerging priorities on an ecosystem. This concept of emergence means that properties cannot be predicted from a knowledge, even in detail, of the biology of each species and do not result from the simple addition of activities belonging to each species. Without using the term "super-organism"[2], it is clear that the whole biological population of an ecosystem (its "biocenosis") makes up an integrated original biological system[3]. In addition to the need to describe this new level of organisation of living things ("ecological diversity"), **this observation leads to the need for global conservation of those co-adapted sets that are the biocenoses within their ecosystems and not only conservation of their components independently and outside these ecosystems.** To use an analogy, we must describe not only the instruments of the orchestra, but also the music that it plays, and the idea of destroying all the musical parts in the world on the pretext that we know the notes belongs to the nightmare of *Fahrenheit 451*.

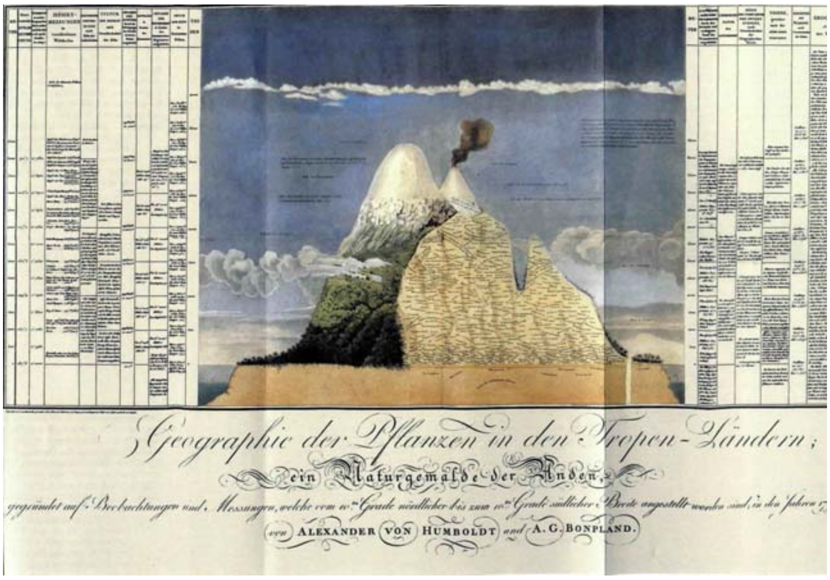
---

[1] The examination of genetic diversity between individuals of the same species and any role of this diversity has provided little motivation, until recent times, for functional ecology, even if Haeckel, the inventor of the word "ecology", was close to Darwin.

[2] Even if the Rio definition allows it to be thought that ecosystems are organisms ("variability of living organisms, including, among others, ecosystems…"), it appears to us that the term should be reserved for extremely close, quasi-indispensable associations between species, like symbioses (such as lichens, which are associations of algae and fungi).

[3] In particular because, due to co-adaptations, the same species in two different ecosystems could have distinctly different biological characteristics, for example in terms of resistance to diseases.

Figure IV-1: Distribution of vegetation on the flanks of the Chimborazo volcano (map published in 1805 by A. Von Humboldt and A. Bonpland)




*Source: commons.wikimedia.org*

Finally, the third concept leads to reviewing the **limits in the use of species as a metric of biodiversity.** In effect, because functional ecology aims to identify the role played by each species in an ecosystem, it is led to group together species playing a similar role, for example, if we study the cycle of organic matter, primary and secondary producers, predators, decomposers, etc. These "functional groups" can gather together species that are very close in evolutionary terms (for example the different species of woodpecker in a forest) or on the other hand belong to very different evolutionary lines (for example grasshoppers and sheep, two grass eating species on grassland)[1]. Another proposition of functional ecology, the idea that a certain redundancy between species often exists in a given functional group, in other words that, within a group relatively rich in species, the increase in the number of species or the disappearance of one of them will not have a notable influence on the operation of the ecosystem and the ecosystem services that they provide; in contrast, if a key function is only provided by a single species, its disappearance must on the contrary be considered as a major loss of biodiversity. **We thus see that this functional approach will take the diversity of species in an ecosystem into account in very different ways and will offer a measure of biodiversity that could be very different from that founded on evolutionary divergences.**

## 1.5. From the description of entities to the measurement of diversity: the perception of fragility

The necessity of apprehending biological diversity not only through a qualitative description of its entities but by a global quantitative measure has appeared relatively

[1] Conversely, species that are close in evolutionary terms could have very different functional roles: plankton filterers, grazers, invertebrate consumers, even predators of other fish among will be found fish species in the same family.

recently. Note immediately that the term “diversity” has no precise sense in the mathematical vocabulary, as a result of which there are varied possibilities for translation in terms of indicators.

This development can be considered as part of a general trend of sciences to want to quantify their approaches, but also, in the case of biodiversity, to more practical preoccupations. So, the precursor work of the Dane Johannes Schmidt at the start of the century[1] on the measurement of the variability of the number of vertebra between individuals in the same species of fish aimed to find out if the variation observed between different environments was of genetic origin and thus meant that there were different stocks (which could thus be managed independently), or resulted from the local influence of the environment on a single stock. Furthermore, these goal oriented preoccupations, in particular those of the genetic improvement of animal races and plant varieties, motivated the development of the statistical methods used to estimate the genetic variability of certain characters from the observable diversity (“phenotypic”) of these characteristics.

As far as species diversity is concerned, the first proposals for global measurement can be traced back to 1948, based on either information theory (Shannon-Wiener index) or on probability theory (Simpson’s index). In both cases, the index is higher if the number of species is higher and their proportions are balanced. But these indices were hardly known outside the specialised scientific community and served mostly to compare the biodiversity of homogeneous sites in the same ecosystem or different ecosystems. It is no doubt the perception since the 1960s of the accelerated erosion of biodiversity, at its different levels of organisation (loss of genetic diversity, through the disappearance of traditional domestic races and varieties or the reduction in numbers of wild populations, disappearance of species, destruction of ecosystems) that has motivated the efforts to construct simple indicators, that can be understood by political decision makers, and, more widely by the general public. Accepted on the evolutionary scale, the disappearance of species became a worrying reality on the human scale. It was no longer a matter of comparing different ecosystems in space but being able to monitor, on wider and wider scales (ecosystems, landscapes, eco-regions), the time variations of biodiversity. The following paragraphs show that indicators suited to the description of the spatial variation of a supposedly stable biodiversity then show themselves to be unsuited to these new terms of reference.

In addition, given the importance of human activities in this erosion of biodiversity, it seemed necessary to define not only status indicators, that can be used to monitor changes, but also indicators of the pressure (or interaction) of these activities on biodiversity.

## 1.6. Conclusions on biodiversity concepts

After this general survey, we can summarise the concepts that biodiversity indicators must translate, but also the constraints that are imposed on these indicators:

- Report, from a necessarily limited number of easily observable entities, on a much larger whole, that is still largely unknown.

[1] See for example *Journal of Genetics*, 1919.

- Describe the different levels of organisation of biodiversity (genetic, species, ecological) by using, at least today, specific metrics for each level without common features.
- Go beyond the inventory of entities to take account of the importance of interaction between them, whether in the short tem as the foundation of ecosystem services or in the long term as the motor for the adaptation of the living world.
- Appreciate and measure, on a human scale, any variations of this biodiversity and the evolution of the factors responsible for these variations, in particular human activities.

**It is thus clear that biodiversity is a "multi-dimensional object", whose different dimensions additionally do not obey common metrics.** We can thus already state that it is illusory to hope to describe it by a single indicator. **We could also be tempted to speak of *multiple* biodiversities, in so far as the different levels, intra-species, species and ecological, are not currently the subject of a single theory that can be used to group them together. However, we will continue in the rest of this report to talk of *the* biodiversity, because, even if the appropriate concepts are still partly to be built, it is evident that it involves speaking of the deployment, in time and space, of living things controlled, beyond their diversity, by a common origin and processes.**

## 2. Options for the quantification of biodiversity

Biodiversity indicators offer *"the opportunity to create bridges between the world of experts and the world of the layman, between that of science and that of politics, facilitating the emergence of a common language about this subject called biodiversity"* (Levrel, 2007).

From a scientific viewpoint, they contribute a synthesis of the knowledge of the state of biodiversity for the quantifying of general phenomena but with results that must be interpreted with care. From the point of view of society, they must be understandable by everyone, so as to establish a dialogue and compromise between the stakeholders.

### 2.1. What should be measured?

#### *a. Different units for specific ecological and institutional contexts*

In the current context of accelerated environmental modifications, where anthropogenic modifications propagate throughout the biosphere, **biodiversity comes down to a great diversity of conservation objectives** supported by a great diversity of players (Barbault and Chevassus-au-Louis, 2004). Introducing indicators (of ecosystem resilience, species richness, genetic variability or quantity of ecosystem services) necessitates preliminary fixing of precise conservation objectives, in close relationship to the way in which the players concerned will use it).

Monitoring biodiversity cannot then be summarised as a limited number of taxonomic (common birds for example) or structural indicators (species composition for example). If these common indicators can be use to produce comparisons, it is equally interesting to develop **indicators that are adapted to specific ecological and institutional contexts**.

Their construction currently involves tradeoffs between the tensions that will represent the scientific and political bases of the indicator (Couvet *et al.*, 2004; Levrel, 2007):

- The double political and scientific dimension associated with the indicators: implies the provision of simple information to a wide public or a tool whose results must be interpreted with care (limits the simplicity),
- The scale of application of the indicator: an indicator is thought of on specific spatial, time and symbolic scales, which makes it difficult to transpose to another scale,
- The subjective and objective character of the indicators: even though indicators are biased and partial information tools, they encourage public discussion. Bringing them into question is useful but can lead to a loss of confidence in this tool for argument.

### *b. A complex object*

The quantification of biodiversity, combined with a double scientific and social dimension, is **an objective that is as ambitious as biodiversity is a complex object**, in both its different levels of organisation (genetic diversity, species, and ecosystem) but also because of the heterogeneity of the entities within each of its levels. The complexity of the object to be measured and its perceptions (by ecologists, taxonomists, naturalists, managers, economists, etc.) causes a large dispersion of representations and information about it (Levrel, 2007).

The complexity of biodiversity leads to a multiplication of methods of construction and choice of biodiversity indices. The indicators refer to numerous spatial (of damage to the biosphere), time (non comparable genetic, species or ecosystem changes) and symbolic (representations) scales. Similarly, depending on the decision-making or functional angle from which biodiversity is approached; we can distinguish decision-making units (communes, cantons…) or functional units (hydrological functioning of drainage basin units for example).

An indicator is thought of on specific spatial, time and symbolic scales, which makes it difficult to transpose to another scale. Developing a realistic biodiversity indicator for a scale and for the use of specific players (scale of realism or application), is to admit that this indicator will probably be unrealistic on another scale and unsuited to the representation of the players who use it. So, the objective of national scale is to show the efforts made to achieve the objective of a reduction in erosion of biodiversity by 2010 and the results obtained. On the local scale, the biodiversity indicators used by managers more often respond to development objectives for their area. As for the general public, it is sensitive to certain species with which it associates representations (bear, wolves, deer, lynx…) and/or that are easily observable (birds, common butterflies…) (Levrel, 2007).

The same problem of scale affects the time dimension; the decision makers do not think in the same time steps. Conversely, more precise biodiversity indicators present limits of use on a large scale.

The multidimensional character of biodiversity thus represents a real challenge for the construction of a metric taking account of all the dimensions of biodiversity (diversity of species, functions, interactions, genetics etc.).

### *c. A biodiversity sample*

A biodiversity indicator is constructed from data that is itself only a sample of the biodiversity represented. **The existing indicators thus only provide a partial approach to current changes in biodiversity.** To monitor and analyse these changes in greater detail, and at their different levels, we need to set up multiple extended observation systems, that can be used to compare the status of biodiversity in all spaces, at different scales and under different pressures (to identify the factors responsible for variations).

When we examine the indicators that are currently used (see below) we note that these indicators are based on variations of the abundance of group of a species chosen, depending on an ecosystem type, a geographic zone, but **none are strictly based on the diversity.**

## 2.2. What is the ideal indicator?

The construction of biodiversity indicators responds to an increasing need to have pragmatic information tools responding quickly (consistent with the rate of modification of the biosphere) and unequivocally compared to human actions.

### *a. Ideal properties*

Whether it relates to species, intra-species (genetic) or supra-species (ecological) diversity, a good indicator must take account of:

- The richness, meaning the number of different entities present.
- The equality between these entities. For the same number of entities, it is legitimate to consider positively a balanced distribution compared to a distribution with some dominant entities and numerous very minor entities.
- The diversity, meaning the “distance” (or the relative originality) of these entities. This concept of differentiation can be viewed in evolutionary (phylogenetic distance) or functional (ecological role) terms, with these two concepts leading to very different approaches. In different biomes the same functions are often found to be exercised by species with wide evolutionary separation. So the marsupial fauna of Australia includes predators, rodents or herbivores, and appears functionally very similar to European fauna, although they are very distant in evolutionary terms.

In addition, the indicator must be:

- Able to report the absolute abundance of species or populations: this concept of absolute abundance is not always associated with the concept of biodiversity, but it appears indispensable to include we want to take account of the role that it plays in the functioning of the planet in the concept of biodiversity, and in particular, to link the concepts of biodiversity and ecosystem services (developed below). From this viewpoint, it is legitimate to consider that the reduction in abundance of a given population is a loss of biodiversity.

- At ecosystem level, able to include the spatial organisation of entities (connectivity, distance, position in relation to the topography), which we know plays a major role in the characteristics of the whole (landscape ecology).

### *b. The necessity of composite (or multiple) indicators*

**It is impossible to define a single indicator accounting for all the aspects of biodiversity. In addition, synthetic indicators can mask important realities.** We will take the example of two "groups" (populations) (table IV-1) containing a variable number of "biological entities". These entities can be species, or at the genetic level, alleles, meaning different "versions" of the same gene. If we evaluate the "richness", meaning the number of different entities, we will say that group A, which contains six entities, has a higher biodiversity than group B, which only included three. If we take the relative abundance into account, we can measure the diversity by calculating the probability of drawing two different entities by taking two entities from the group at random. This probability is 46% in group B and only 19% in group A, composed essentially of a single entity. We could thus say that biodiversity is higher in group B.

**Table IV-1: Comparison of the biodiversity of two groups: frequency of different entities**

| Entities / Groups | 1 | 2 | 3 | 4 | 5 | 6 |
|---|---|---|---|---|---|---|
| A | 0.90 | 0.02 | 0.02 | 0.02 | 0.02 | 0.02 |
| B | 0.70 | 0.20 | 0.10 | | | |

*Source: B. Chevassus-au-Louis*

So what is the "correct" approach? In fact, everything depends on what you want to estimate:

- If we are interested in the biodiversity at a given moment, the second reasoning is correct: species diversity is better in group B, because all the species are present with notable frequencies and will thus be able to play a significant ecological role. Similarly, if genes are considered, group B will include a higher number of heterozygous individuals, meaning two different alleles of the same gene, and we know that this heterozygosity is often linked to better performance and better individual adaptability;
- In contrast, if we are interested in long term evolutionary abilities, group A has better potential: depending on the environmental pressures, some of the numerous entities (species or alleles) that it contains could reveal themselves suitable and increase in frequency, whereas group B has a more limited "hand".

**Therefore, having multiple indicators is not only inevitable but also necessary. As in human health, defining the "state of health" of biodiversity cannot be done by measuring only one parameter. This does not prevent us from expressing, after**

**considering different indicators, a general judgement on the fact that the biodiversity is effectively "in good health".**

## 2.3. The difficulties of constructing a general biodiversity index

### *a. Choices of simplification, aggregation, weighting*

To identify a particular dynamic associated with a habitat we have to, on the one hand, take a large range of species in that habitat into account so as to return the complexity of the system being studied, and on the other hand, compare it with indicators in other habitats to bring out the particular character of the indicator's dynamic in the habitat considered. In such a multi-species indicator, **it will not necessarily be a case of adding the data for a maximum of species, but of selecting them and combining them whilst giving them an individual weight depending on the objectives of the indicator.** The combination must be relevant in the light of the functioning of the ecosystems.

Whatever the measured biodiversity characteristic – genetic, species or ecosystem diversity –, the construction of a general biodiversity index is thus based on a set of scientifically motivated choices: 1) the choice of the method of combining the populations or species, and the criteria for assessing the state of these groups; 2) the choice of the method of weighting the species and/or groups (Couvet *et al.*, to be published).

In the current state of knowledge, the basis **for richness** will be:

- At the genetic level: the diversity of some easily locatable genes (microsatellites, SNP), whilst waiting for the sequencing of complete genomes for individuals in the population. We thus define the "allelic richness". The question of how representative these genes are compared to the whole genome is arguable because we currently know that different regions of the genome can be more or less rich in diversity;
- At the species level: on known and "macroscopic" species, to allow easy inventories (vertebrates, higher plants, emblematic insects) assuming that they demonstrate a more general biodiversity.
- At the ecological level: we will mainly base it on plant populations and their associations (phytosociology) to define habitats. These habitats will be the elementary unit of the inventory. The concept of an ecosystem remains complex and poses problems of spatial limits on the ground.

An important problem at the first two levels is the sensitivity to the sampling effort. It is clear that the possibility of finding a rare allele in a population will increase with the number of individuals studied; idem for species. Therefore, we must develop models called "saturation curves" to find asymptotic estimates of richness.

**For equality**, the relative abundances of the entities identified can be weighted, assuming that there is no bias (some species are more difficult to trap or observe than others).

**For** species and genetic **diversity**, information on evolutionary distances is often not available, although it is known that certain entities can be very close (twin species) and others very distant. Here too, genomic approaches will allow objective measurement of distances but are not operational. For ecological diversity, it is equally difficult to measure a “distance” between habitats. This dimension is thus not generally taken into account and entities are implicitly considered as “equidistant”. In contrast, on the functional level, species can be combined into “functional groups” (for example in terms of positioning in the food chain) and the diversity of functional groups measured.

**For absolute abundance**, we will generally be limited to some easily locatable groups for which demographic data can be obtained. Here too, we will assume that, taking the multiple interactions within the ecosystem into account, these abundance fluctuations represent more general changes (see STOC indicator).

Finally, for **spatial organisation**, the definition of a general parameter accounting for the spatial distribution of habitats and having a functional meaning remains a challenge. A ratio could be proposed between the area (hectares of forest, ground occupation area according to CORINE Land Cover…) and the quality of these systems evaluated relative to a reference value.

### *b. Case of unmeasurable concepts*

The construction of a biodiversity status index from unmeasurable concepts can be illustrated by the use of the calculation of the “biodiversity integrity” index (Scholes and Biggs, 2005 *in* Couvet *et al.*, *to be published*).

The index is calculated by weighting the observed abundance of each functional group by its diversity in number of species and by the area of the ecosystems observed. It combines measurements obtained on different groups, according to different spatial configurations and so on different spatial scales.

In countries where there is not sufficient information to make calculations of the relative abundance of populations, the indicator proposes an approximation of the change in biodiversity from an estimate, by an expert, of the impact of human activities on reference animal and plant populations and generalises this impact to all the populations belonging to the same functional groups. The impact is estimated on the advice of a minimum of three specialists for each taxonomic group (plants, mammals, birds, reptiles and amphibians).

This indicator also allows monitoring from ecosystem scales, from activities having an impact on habitats or from functional groups.

**In a context where many scientific uncertainties persist, where the availability of data is a limiting factor in the development of indicators, we should give priority to modest and prudent approaches for the conceptualisation and use of biodiversity indicators.**

The setting up of biodiversity observatories should gradually free the documentation of biodiversity indexes from the advice of experts.

# 3. The main types of biodiversity indicators

The previous paragraphs have brought out the complex character of biodiversity and the multiplication of choices of associated biodiversity indexes, referring to numerous spatial scales. The construction of a general biodiversity index is an even more difficult challenge because, as well as assessing and monitoring the state of biodiversity, the indicators now also have the ambition of improving its management. In fact, in the current context of accelerated anthropogenic modifications, **we need to have an integrated approach to species management taking account of inter-species relations and evolutionary processes, ensuring the durability of the adaptation process for natural systems.** The biodiversity indicators used during biodiversity consultations and the conservation strategies must thus be clear and relevant to facilitate any arbitration between different socio-economic interests.

The indicators selected and presented below are indicators developed by France that have the benefit of a data collection and analysis organisation. The indicators selected in 2007 by the European Environment Agency (EEA) under the Convention on Biological Diversity (CBD) are also described.

By proposing different biodiversity indicators (abundance, diversity, ecosystem functional status, or more precisely, the biocensosis – endemic rates, mean trophic level, specialisation rates, etc) –, we are attempting to emphasise the multiple dimensions and are placing ourselves in a situation comparable to that of economists, who set up a diagnosis from several indicators.

## 3.1. Status indicators: abundance *versus* species diversity

A first category of indicators proposes the measurement of biodiversity from a single parameter (the species, the individual, the gene, the interaction), which implies very precise and exhaustive knowledge of the system. In the absence of sufficient data, we are content with more directly measurable characteristics of the system's state: the abundance (number of individuals) and the species diversity.

If the species richness is currently considered as uninformative (insufficient knowledge of taxons, variable responses to taxons, low sensitivity to short term variations, etc.), in contrast, **the abundance indicator presents the advantage of being sensitive to short term dynamics and constitutes a relevant indicator for assessing the state of health of the ecosystem** (Levrel, 2007).

Several arguments plead in favour of biodiversity monitoring based on an estimate of abundance variations: i) species abundance variations can be interpreted by including the different mechanisms controlling their evolution; ii) these variations are quicker and more continuous than variations of species diversity; iii) species diversity variations can be derived from them, but the converse is not true.

An index of the ecosystem's state can be derived from these abundance variations by calculating a global index weighted according to the trophic level (Pauly *et al.* 1998 *in* Couvet *et al.*, 2004). We can also go further by distinguishing individuals according to their age, size…), or by estimating demographic (survival, reproduction success) or genetic (consanguinity) parameters, which allows the diagnosis of the evolution of these populations to be refined.

**A whole set of indicators can be derived from the results of the abundance monitoring** and can be classified according to three objectives:

### *a. Describing the biodiversity dynamic*

Describing the *biodiversity dynamic* (erosion, stable state, etc.): According to different habitats (agricultural, forest, heritage interest habitats, wetlands, etc.), different species groups (according to their conservation status, trophic level, area of origin, etc.), different ecological services (carbon storage, soil fertility, pollination, disturbance control, etc.).

Two indexes can be mentioned under this heading, the "Red List Index" (RLI) proposed by the IUCN (International Union for Conservation of Nature, now the World Conservation Union) and the "Living Planet Index" (LPI), developed jointly by the UNEP (United Nations Environment Programme) et le WWF (World Wildlife Fund).

- The RLI measures the mean variation of conservation status for the species on the red list. It is based on a classification of species into several categories, running from "not threatened" to "extinct" passing through "vulnerable", "threatened" and "critical". This classification, applied to vertebrates, shows for example that set of species considered as "threatened" or "critical" – and so likely to disappear in the short term – is very much greater than that for currently extinct species, in particular for amphibians, reptiles and fishes. The LRI thus allows the measurement of the durability of species, from a heritage viewpoint. The LRI index for birds has fallen by 6.9 points from 1988 to 2004 (Butchart *et al.,* 2004), which translates as a distinct average degradation in the status of threatened species.

- The LPI measures the global variations of the number of vertebrates on the planetary scale. It is based on the estimated numerical abundance of 3,000 populations representing 1,100 species of vertebrates. Calculated since 1970, it may have fallen, since this date, by 30% for land and marine species and 50% for fresh water species. The analysis possibilities are limited by the arbitrary choice of populations monitored and the few species considered by space and functional group.

### *b. Quantify the pressures*

Quantifying *the pressures* bearing down on biodiversity and *the effectiveness of the responses* provided (PSR model, OECD 1994) by comparing the dynamics of functional groups according to habitats: estimating the different pressures (impact of homogenisation on agricultural landscapes, urbanisation, agrarian practices or forest management methods, climate warming, etc.), and the identification of the most relevant responses (Natura 2000, agri-environmental measures, different levels of protection of spaces…).

### *c. Measuring the closeness of objectives*

Measuring the *closeness of different objectives* (stopping the erosion of biodiversity in 2010, maintaining species and habitats of community interest in a favourable conservation state…).

## 3.2. Indicators of the state and functioning of an ecosystem: weighting of species indicators by their biological characteristics

The concept of biodiversity is often reduced to that of indicator species (also called bio-indicators)) (Couvet *et al.*, 2004). The presence of these species is used to characterise the "quality" of an environment. However the concept of biodiversity goes further than the simple description of the diversity of living things, even if it were exhaustive (Barbault et Chevassus-au-Louis, 2004), by describing the interactions at each functional level between the functions but also with human societies. The complexity of dynamics that drive biodiversity is however not or poorly accounted for by indicator species presence-absence data.

An ecosystem includes numerous interconnected functional groups, whose abundance variations are not necessarily parallel. However to evaluate and monitor the state of an ecosystem, and above all to communicate clearly with the general public and decision makers, we may wish to summarise the data in a single index, including the variations in abundance of very different systematic groups such as mammals, insects or plants.

Single parameter indicators (species, individual, gene, interaction) provide targeted information, with little integration, and require very precise knowledge. Composite indicators, for their part, imply the use of at least two different reference units. **The approach using composite indicators, without being a miraculous solution, offers three advantages:** 1) It reduces the stochastic problem by an averaging effect; 2) It allows targeted information by grouping species using the same functional criterion; 3) It offers a common reference unit that eases interpretation and limits the weighting problem (example: common birds) (Levrel, 2007).

However, composite indicators are confronted by two recurrent questions (Couvet *et al.*, 2007): the data aggregation and weighting methods. **Species groupings** can be done according to the systematics, the functions provided within an ecosystem (example: the trophic level), the method of use by man (fishing, hunting, ecotourism) to ease the interpretation of the observed variations. **The** simplest **weighting option** is to assign the same weight to each species (example: monitoring birds, butterflies, mammals and plants). A second "conservationist" option consists of weighting the weights of species as a function of their rarity, the threats of extinction that bear on them or their symbolic character. The last "ecological" approach assigns a superior weight to species fulfilling essential ecological functions[1] (Couvet *et al.*, 2007).

Depending on the option chosen, we arrive at an index of species richness, function or rarity.

[1] Useful ecological processes that result from interactions between species and between species and the environment. Examples: water quality maintenance, climate regulation.

For the choice of species, the indicators linked to threatened species are unavoidable in judging the achievement of objectives defined in relation to these species and are well reported. However, **the indicators associated with common species are being developed** to go further in the understanding of the mechanisms. The future of these species is important because they are indispensable for correct operation of the whole ecosystem, consequently to the provision of numerous ecological services. In addition, from a methodological viewpoint, they present the advantage of being widely distributed, which allows the sampling of a great diversity of habitats and the separation of the effects of habitats and protective measures, by sampling both protected and unprotected spaces. The variations in their numbers, which are high by definition, are more easily interpretable than those of rare species, subject to random variations.

### *a. The Communities (C) Index*

This index is used to measure the ecosystem's functionality by describing, for a given character, the mean state of a biocenosis through the weighted mean of the values of each species. The species' character can be morphological (size), characterise the life history, the durability, the number of propagules… the ecological requirements of the species (thermal, humidity), the habitat's specialisation or trophic interactions, or again the trophic level.

The calculation formula translates the importance of integration different parameters to explain the complexity of the biodiversity object:

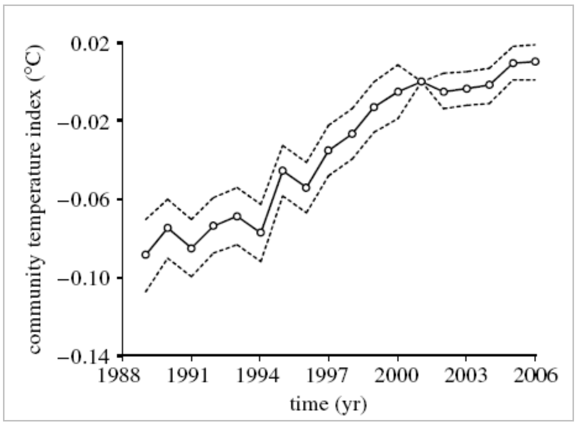
$$C_{kj} = \sum_{i=1,n} \frac{N_{ik}}{N_k}.S_{ij}$$

With:

- *Sij*: mean value of the character *j* for species *i*
- *Nik*: number of individuals of species i in community *k,*
- *Nk*: total number of individuals in community *k*,
- *Ckj*: value of character j in community *k* *(community index)*

So in the communities' thermal index, the species can be characterised by their mean thermal preference for temperatures in their distribution area. We can then calculate a "thermal index" – mean of abundance weighted species' thermal preferences – and analyse its variations, notably in response to climate changes. For example, the bird communities' thermal index (CTI), calculated throughout France, has demonstrated the increase in the number of southern species during the last twenty years, due to climate warming (figure IV-2) of 50% compared to the expected increase (obtained by the index-temperature correlation for the sites) (Devictor *et al.* 2008).

**Figure IV-2: Variation of thermal index for bird communities throughout France between 1988 and 2006**




The CTI being the mean of each STI (index for a given species) weighted by the abundance of the species.

*Source: Devictor* et al.*, 2008*

Another example, the marine trophic index (MTI = *Marine Trophic Index*) allows evaluation of the **state of marine food chains** in the various great ocean regions. The indicator is calculated from catches declared by fishermen, generally recorded at national level: if the species captured is a consumer of plant species (phytoplankton, algae), it will have a trophic level *T* of 2, if it consumes plant eating species, its trophic level will be 3, etc.

$$MTI = \sum_{j=1}^{n} M_j T_j$$

With:

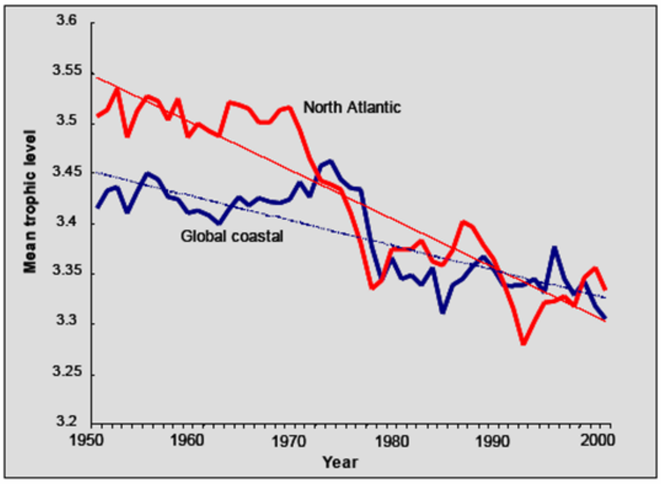
*Mj* represents the relative abundance of species *j* in the catches,

*Tj*: the trophic level of species *j*.

Figure IV-3 shows the clear reduction of this indicator during recent decades, in all oceans. It translates the development of fisheries towards species situated lower and lower in the food chain – and thus generally of less economic interest –, as a consequence of the reduction of predators situated at the top of the chain.

Application to the **terrestrial food chain** also shows perturbations. For example, the uncontrolled proliferation of cervid populations causes major damage to agricultural areas and wild plants and blocks the regeneration of forest environments.

Figure IV-3: Variation of the marine trophic index in the North Atlantic (in red) and throughout coastal zones (in blue) between 1950 and 2000




*Source: Pauly and Watson, 2005*

### *b. The "common birds" indicators (STOC programme)*

The "common birds" indicators are **used as communication and decision making tools,** on both national (by the IFEN) and European scales (by the EEA). The indicator of variation of abundance of common birds represents the only biodiversity indicator among the 45 sustainable development indicators for France (IFEN, 2003) and is one of the 15 key sustainable development indicators for the European Union.

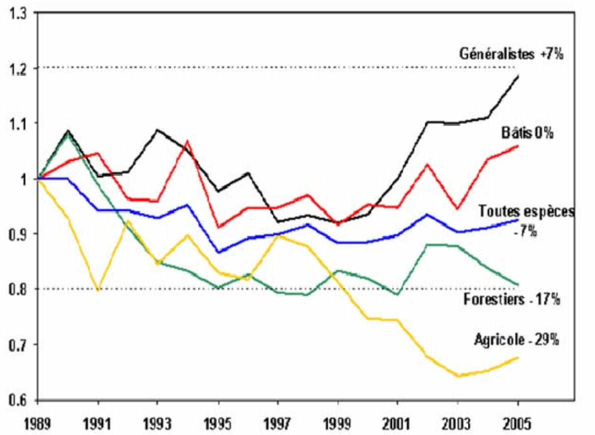
These indicators have been successful with public opinion, scientists and decision makers. They have the advantage of resolving three major problems that all indicator designers must face: the costs caused by this construction, the difficulty in transmitting signals that make sense for a large variety of potential users and the scientific rigour on which the indicators must be based.

Based on the populations that contribute most to the functioning of ecosystems and their development (contrary to rare species), **they are effective tools to evaluate ecosystem operation.** In addition, being situated at a high level in the food chain, common bird populations are indirectly sensitive to disturbances of ecosystem components and **offer an indication of the state of health of ecosystems**. Another non negligible advantage is that they are based on monitoring populations whose size is very sensitive to short term environmental changes, so supply effective policy evaluation tools to measure advances related to the 2020 objectives (Levrel, 2007).

The indicators are developed from the STOC (Time monitoring of common birds) database, which contains demographic information such as the abundance, distribution, diversity, fertility, and survival of common bird populations and communities. The monitoring systems allow **the assessment of ecological dynamics at both fine and large spatial scales.**

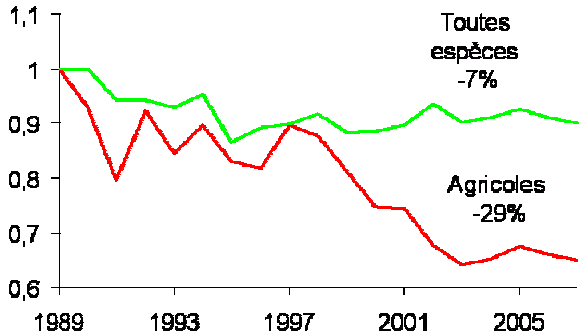
By characterising the state of a group of species associated with a particular environment, the specialist species indicators clarify the state of health of particular environments. So, on the European scale (figure IV-4 and figure IV-5), the results show a general decline of specialist species in different environments. This decline is particularly high in French agricultural areas where the variations in numbers of the "agricultural birds" indicator show a decline in the specialist species of these environments. This result is also observed in other comparable agricultural countries.

Figure IV-4: Abundance variations for common birds (95 species) in France by habitat (statistics prepared from over 5 millions individuals observed or captured)




*Source: IFEN Internet site*

Figure IV-5: Abundance variations for common birds in Europe, average of 18 European countries by habitat

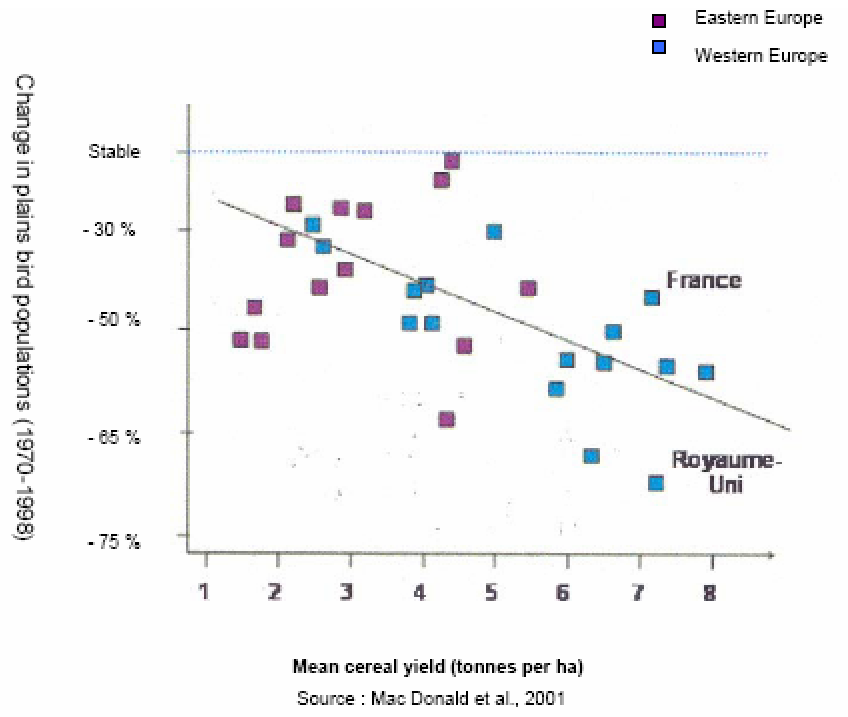



*Source: Eurostat Internet site*

## 3.3. Pressure indicators

The intersection of state indicators with other indicators, arising from related observation systems that supply information on activities that could influence biodiversity, allows us to go further in understanding the causes of variation.

Figure IV-6: Correlation between the abundance of plain birds and agricultural production in different European countries




For example, the correlation with agricultural production (figure IV-6) shows a decline in the populations of plain birds that is well correlated with the countries' agricultural performance (Donald *et al.*, 2001).

To go further into this question of global changes affecting habitats, an indicator has been developed and allows the change in the degree of specialisation of common bird communities to be linked to the degree of fragmentation and disturbance of their habitats: the Communities Specialisation Index (CSI).

### *a. The Communities Specialisation Index (CSI)*

The Communities Specialisation Index (CSI) provides a ratio between a number of individuals from so called specialist species, which possess a narrow ecological niche, and a number of individuals from so called generalist species, which are less demanding[1]. It provides an estimate of the degree of specialisation of species from a coefficient of their density variation by habitat, identified from the CORINE Land Cover soil occupation directory (Julliard *et al.*, 2006).

By examining the correlation between this level of specialisation of communities (figure IV-7) and two other indexes measured independently – the degree of habitat fragmentation and the degree of disturbance – we can observe the link between the specialisation index and the state of the habitat.

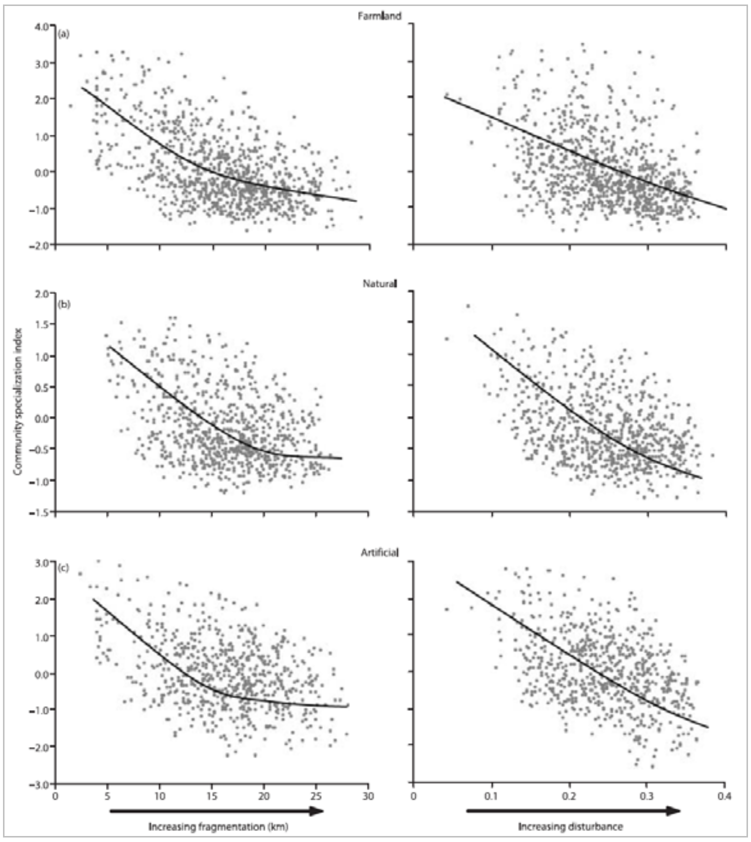
[1]When there are significant modifications of habitats, specialist species, which theoretically evolve in stable environments, often see their numbers diminish whereas generalists are more often favoured because they are less demanding.

Figure IV-7: Change in specialisation index with change in fragmentation (on the left) and disturbance (on the right) of soil occupation (CORINE Land Cover coverage).

Fragmentation

(Soil occupation heterogeneity) Disturbance

Soil occupation change rate

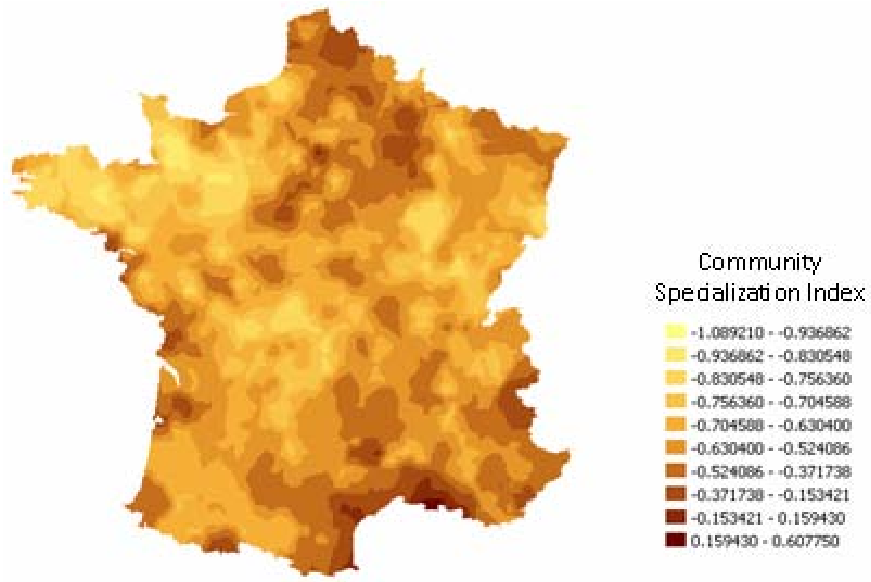



Ordinates: relative value of the CSI. Abscissae: the fragmentation index (on the left), the disturbance index (on the right)

*Source: Devictor et al., 2008*

So we observe (figure IV-8), for different habitat types (agricultural (a), natural (b), and urban (c)), a reduction in the number of specialist species with increase in habitat fragmentation and disturbance. This negative correlation is also observed for biogeographical zones.

Figure IV-8: ISC mapping (mean between 2001 and 2007,
from 10,000 observation points)




*Source: Community Specialization Index (Jiguet, unpublished)*

## 3.4. Social preoccupation indicators

**Beyond the assessment and monitoring of biodiversity status, indicators have the role of improving its management**, taking account of multiple constraints related to space and available resource limitations. All biodiversity management must analyse the biodiversity dynamic at large and large spatial scales, where ecological and evolutionary processes are interwoven, affecting both common species in human affected areas and threatened species in protected areas.

In this, biodiversity indicators are **a preferred tool to identify these** medium term **biodiversity management strategies.** They must thus be clear and relevant to throw light on arbitration between different socioeconomic interests. The complexity of biodiversity (the object and its representations) must not be underestimated in any way, on penalty of starting development and conservation policies that will lead to mixed, or even catastrophic results.

Rather than the traditional name of “sustainable development indicators”, we will use the interesting name of “interaction indicators” proposed by Levrel (2007) to describe the interactions that exist between biodiversity dynamics and socio-economic dynamics. This category of indicators is intended to apprehend the socioeconomic dynamics that give rise to the threats experienced by biodiversity. Their construction implies **a compromise between three fundamental tensions: its contextual and universal, scientific and political, arbitrary and pragmatic dimensions.**

### *a. The CBD society-nature indicators*

Under the **Convention on Biological Diversity** (CBD), 13 indicators of interactions meeting the double scientific and social criteria were selected in 2004. These generic indicators are intended to evaluate the progress accomplished globally in the pursuit of the objective of ending the erosion of biodiversity by 2010. Some concern the direct impact on biodiversity; others reflect attacks on biodiversity, its sustainable use and its integrity. The set of indicators allows assessment of the effect of different sectors and sector policies on biodiversity.

- Four of these indicators are biodiversity status indicators. In addition to the three that we have already defined (the Red List Index (RLI), the Marine Trophic Index (MTI) and the Common Birds Index (STI, STOC in France), the CBD proposes the integration of a measure of the genetic diversity of species used by man (raised, cultivated or exploited).

- Five indicators relate to the occupation of space and the quality of environments: land occupation, coverage of protected areas, areas benefiting from sustainable management, ecosystem connectivity/fragmentation, water quality in aquatic ecosystems.

- Two indicators relate to anthropogenic pressures on ecosystems: importance of atmospheric nitrogen inputs; number and costs of biological invasions.

- Finally, two socio-economic indicators are proposed, related to linguistic diversity and States' financial support for the CBD.

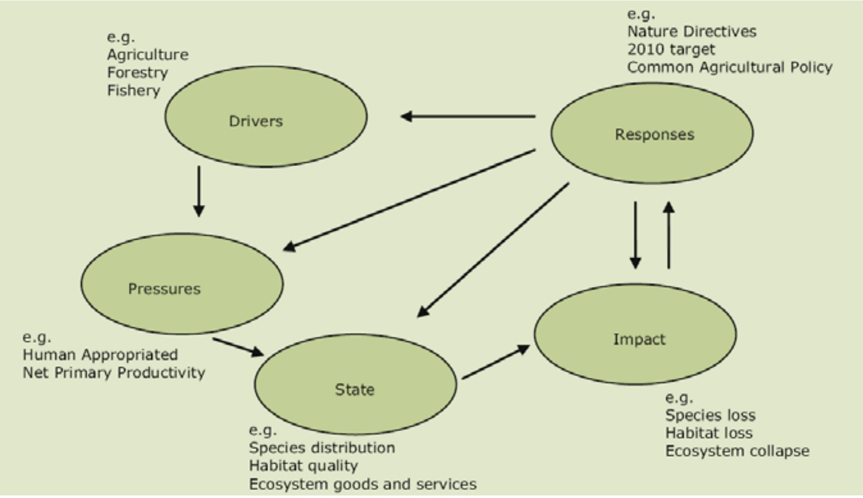
*b. The "Streamlining European Biodiversity Indicators (SEBI 2010)" programme*

To harmonise the initiatives undertaken by the European CBD member countries for the definition of indicators of progress towards the 2010 objective with those taken at the European Union as well as the global level, The European Environment Agency (EEA) has coordinated the implementation of the Streamlining European Biodiversity Indicators (SEBI 2010) programme.

Based on 13 generic indicators selected by the CDB, 26 indicators were selected for Europe in 2007, intended to evaluate the effect of the different sectors and sector policies on biodiversity. The aim is to monitor the progress achieved in Europe in pursuit of the objective of stopping the erosion of biodiversity by 2010 (EEA Technical Report, 2007).

The indicators are used to estimate the direct impact on biodiversity, or reflect damage to biodiversity, its sustainable use and/or integrity, in a Driver-Pressure-State-Impact-Response type analysis framework (DPSIR, figure IV-9).

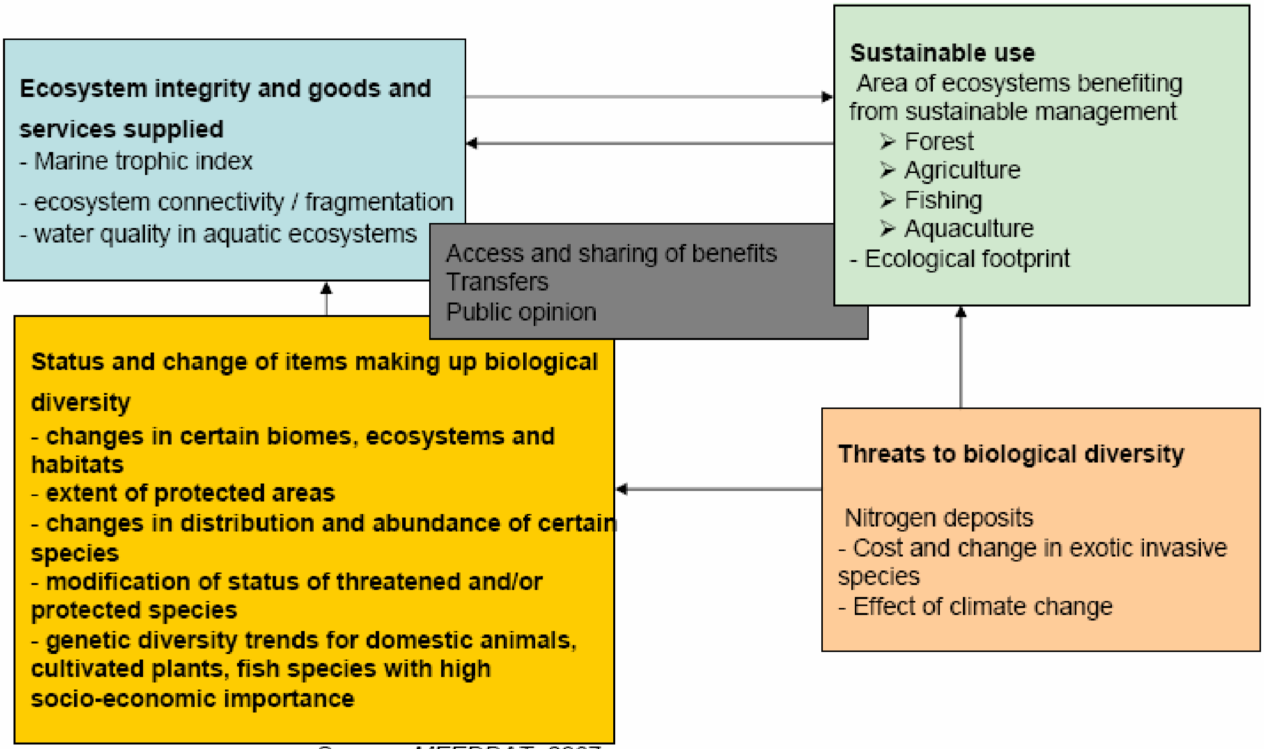
Figure IV-9: Indicators for different DPSIR categories




*Source: EEA, 2007*

Seven challenges have been defined at community level, referred to national level as part of the process of constructing national biodiversity monitoring indicators (MEEDDAT, 2007) (figure IV-10). The French objective is to arrive at a relatively small number of result indicators, used to illustrate the major challenges in a way that is clear and suitable for the largest number of people. As a priority these indicators must be based on already existing data and collection systems.

Figure IV-10: Themes and challenges selected by France, inspired by European work under the SEBI 2010 programme (Streamlining European Biodiversity Indicators toward 2010)




*Source : MEEDDAT. 2007*

The indicators proposed under the European programme (SEBI - Streamlining European Biodiversity Indicators toward 2010) and those selected by MEEDDAT are presented in Table IV-2 for Metropolitan France, and in Table IV-3 for the French Overseas Departments.

**Table IV-2: List of 26 indicators proposed under SEBI for monitoring progress made in Europe (EEA, 2007) and indicators selected by France (MEEDDAT, 2007)**

| Themes | Generic indicators | 26 indicators proposed by the EEA | Indicators selected by France |
|---|---|---|---|
| Status and changes of biodiversity components | Species abundance and distribution | 1. Change in species abundance | Change in abundance of common birds, butterflies, freshwater fish, fished sea fish |
| | Status of threatened and/or protected species | 2. Number of species in IUCN red lists.<br>3. Conservation status of species concerned by Natura 2000, Habitats Directive | Number of species in IUCN red lists.<br>Conservation status of species concerned by Natura 2000, Habitats Directive |
| | Area of selected biomes, ecosystems and habitats | 4. Change in area occupied by the main soil occupation types<br>5. Conservation status of community interest habitats | Change in area occupied by the main soil occupation types<br>Conservation status of community interest habitats<br>Dominance, in the landscape, of environments with little artificial component |
| | Genetic diversity | 6. Number of animal races and plant varieties | Number of animal races and plant varieties |
| | Protected areas | 7. Area in protected areas: global and by type of protected area<br>8. Area of Natura 2000 sites (Birds Directive and Habitats Directive) | Area in protected areas: global and by type of protected area<br>Area of Natura 2000 sites (Birds Directive and Habitats Directive), adequacy of these proposals |
| Threats and pressures | Nitrogen deposits | 9. Exceeding the critical load | Change in water pollutant content |
| | Biological invasions by non indigenous species | 10. Cumulative list of non indigenous species | Number of management plans |
| | Impact of climate change on biodiversity | 11. Occurrence of temperature sensitive species | / |
| | Loss of natural environments (added by France) | | Area annually made artificial |
| Ecosystem quality and functioning | Marine trophic index | 12. Marine trophic index | / |
| | Ecosystem connectivity and fragmentation | 13. Fragmentation of natural and semi-natural spaces<br>14. Fragmentation of river systems | Change in the diversity of non artificial land at local level. |
| | Aquatic ecosystems water quality | 15. Nutrients in transition, coastal and marine waters<br>16. Fresh water quality | Proportion of fresh water bodies in good ecological condition<br>Proportion of transition and marine water bodies in good ecological condition<br>Foliar deficit index<br>Marine trophic index |

| Themes | Generic indicators | 26 indicators proposed by the EEA | Indicators selected by France |
|---|---|---|---|
| Sustainable uses | Area of forests, agricultural, aquaculture and fisheries systems covered by sustainable management | 17. Forests: growth, stock, differential<br>18. Forests: dead wood<br>19. Agricultural systems: nitrogen balance<br>20. Agricultural systems: areas using biodiversity supporting practices<br>21. Fisheries systems: commercial fish stocks<br>22. Aquaculture systems: effluent quality | Forests: area of forests with sustainable management guarantees and proportion compared to the total wooded area<br>Agricultural systems: area under organic agriculture and proportion compared to the total cultivated area<br>Agricultural systems: area subject to agro-environmental measures and proportion compared to the total cultivated area<br>Fisheries systems: percentage of overexploited species |
| | Ecological footprint | 23. Ecological footprint of European countries | / |
| Benefit access and sharing | Benefit access and sharing | 24. Patents for inventions based on genetic resources | Number of patents for inventions based on genetic resources |
| Resource transfers | Financial transfers | 25. Financing directed towards biodiversity protection | Financing directed towards biodiversity protection |
| Public opinion | Public opinion | 26. Public awareness | Public awareness and participation: place given to biodiversity among environmental challenges |

*Source: EEA, 2007, MEEDDAT, 2007*

## Tableau IV-3: List of indicators selected by France (MEEDDAT, 2007) for Overseas Departments

| Themes | Generic indicators | 26 indicators proposed by the EEA | Indicators selected by France |
|---|---|---|---|
| **Status and development of biodiversity components** | Species abundance and distribution | 1. Change in species abundance | Change in the abundance of common land and marine, birds, protected species, marine turtles, cetaceans, vascular plant |
| | Status of threatened and/or protected species | 2. Number of species in IUCN red lists.<br>3. Conservation status of species concerned by Natura 2000, Habitats Directive | Number of species in IUCN red lists. |
| | Area of selected biomes, ecosystems and habitats | 4. Change in area occupied by the main soil occupation types<br>5. Conservation status of community interest habitats | Change in area occupied by the main soil occupation types: forest regions, area of wetlands, mangroves, area of coral reefs |
| | Genetic diversity | 6. Number of animal races and plant varieties | |
| | Protected areas | 7. Area in protected areas: global and by type of protected area<br>8. Area of Natura 2000 sites (Birds Directive and Habitats Directive) | |
| **Threats and pressures** | Nitrogen deposits | 9. Exceeding the critical load | Water quality |
| | Biological invasions by non indigenous species | 10. Cumulative list of non indigenous species | Number of newly established species |
| | Impact of climate change on biodiversity | 11. Occurrence of temperature sensitive species | Sea level |
| | Loss of natural environments (addition to France)<br>Over exploitation | | Natural area / artificial area<br>Offences at borders (CITES)<br>Fisheries pressure |
| **Responses** | Transfers | | Financing directed towards biodiversity protection<br>Number of published articles |
| | Protected areas | | Area in protected areas (global and by type of protected area)<br>Effectiveness of protected areas |
| | Species management and protection | | Number of IUCN threatened species over number of protected species<br>Number of IUCN threatened species over number of species concerned by management plans<br>Invasive species management plans already introduced<br>Monitoring of reports |

*Source: EEA, 2007, MEEDDAT, 2007*

*Note: These tables prepared for the report show the relations between European and French indicators. The original tables are accessible on the MEEDDAT site, in the 2007 activity report on the National Biodiversity Strategy. There is a separation between the Metropolitan and Overseas indicators, the "list of environment and habitat conservation and correct ecosystem functioning related biodiversity monitoring indicators" on the one hand, and the "list of species conservation related biodiversity indicators" on the other hand.*

## 3.5. Indicator development outlook

Numerous indicators are currently available but many do not have the benefit of a data collection and analysis organisation. The development of biodiversity indicators and information systems is continuing and will offer greater possibilities to associate data and develop metrics that can be interpreted on variable scales.

As part of the new scientific strategy, the Institut Français de Biodiversité (French Biodiversity Institute) has reviewed biodiversity indicators and suggested recommendations for their development (IFB, July 2008). The eventual objective is to have a "biodiversity control panel" as an information tool on French biodiversity and its changes, and notably to provide summaries of biodiversity status, the pressures that cause its erosion and the solutions that are applied. The set of indicators proposed by these scientists, based on those of the European Environment Agency (EEA) (see figure IV-11), and the proposals for their development are detailed in an appendix.

Among the biodiversity indicators offering prospects for development, **the abundance and distribution indicator for selected species** (common birds, common butterflies, etc) **is an essential indicator that provides for interesting approaches to different scales of pressures** (diffuse, local). It could be completed by indicators for other ecosystem functional groups (plants, insects, fish, soils, etc.). The avian fauna can also be considered as a functional indicator of ecosystems and offer very interesting indications of:

- The impact of *climate warming* on biodiversity: by measuring the change in distribution areas for common birds and the reproduction period.
- The *global changes in the state of health of habitats:* the common species are grouped as a function of their degree of specialisation with respect to certain habitats (forest, agricultural and built environments) and the change in abundance within these groups translates the changes in different types of habitats.
    - o The status of other trophic levels will allow a much more complete appreciation of the functionality of the ecosystem (Couvet *et al.*, 2007).
    - o At a higher aggregation level, the living planet indicator (LPI) groups together all common vertebrate species on the planet to measure the evolution of the whole biodiversity of the planet according to the different ecoregions.
- The *variation of ecosystem services*: regulation services (predation of harmful species in fields), extraction services (species hunted and consumed by man), auto-maintenance services (seed dispersion), cultural services ("*bird-watching*"). For example, the decline in populations of insect eating birds in agricultural spaces shows the decline of the "biological control" function for insect populations in these environments.

Biggs *et al.* (2004) have developed a **biodiversity integrity index** (Biodiversity Intactness Index - BII), **a promising aggregate indicator** in a context where data requirements are large and strongly associated with particular scales. Based on species abundance, the BII can be used to monitor the abundance reductions related to a certain level of human modification of the environment at different scales, in a specific South African context. The objective is to establish a link between the reaction of species to divers land uses and disturbance thresholds. Changes are identified relative to a reference corresponding to the state of the landscape before alteration by industry.

The BII provides a biological diversity status in a given geographic area, which can equally be a political or ecological boundary. So, in South Africa, it is applicable to the different national, provincial and local decision making scales. It can be disaggregated according to spaces and taxonomic groups, so as to **provide information that can be interpreted by different users**.

In addition, a relation between biodiversity and agriculture intensity was established through estimates of **"high natural value" agricultural areas** (*High Nature Value* - HNV). The HNV indicator offers **an interesting view of the development of quality indicators for a system** through its high spatial and temporal correlation with bird populations. These areas are considered as supporting species and habitat and/or rare species diversity. The comparison between the HNV areas indicator (combination of rotation, extensive practices and fixed landscape component indicators) and the indicator of agricultural bird populations shows a positive correlation between HNV zones and the richness in specialised species. This is consistent with the replacement of specialised species with generalist species with the intensification of agriculture. If there is no linear relation between the total richness of bird species and the HNV, the latter nevertheless constitutes a relevant indicator in determining high natural value areas for the conservation of bird communities.

The ecological character of HNV areas must however be dissociated from that of the agrarian system. The agro-ecological recognition implies inclusion in the instruments of community policies. The concept of high nature value agricultural areas was confirmed in 2003 and a high proportion of HNV areas are now subject to biodiversity favouring measures under rural regulatory instruments. The European Union is introducing measures aimed at supporting agricultural biodiversity and contributing to achieving the objective of stopping the loss of biodiversity by 2010. The conservation of biodiversity depends in large part on the application of measures within the Common Agricultural Policy (CAP), in particular compensation and agro-environmental measures (see the European Commission Internet site). The implementation of remuneration for these agrarian systems remains to be done, notably by mobilising funds for the first pillar of the CAP, article 68: the member states can reserve, by sector, 10% of their national budget envelope intended for direct payments and assign this sum, in the sector concerned, in favour of environmental measures or actions aiming at improving the quality of products and their marketing. The "Grenelle de l'Environnement" environmental round table confirmed the "political and financial recognition for high nature value (HNV) agrarian systems, up to 10% of the national UAA[1]".

In this context, the HNV maps will be a major issue as this theme increases in power in the political field. They will become assessment tools, either *ex-ante* (programming measures) or *ex-post* (judging policies). The validity of such maps will however remain limited to a general view, barely compatible with the assessment of very targeted or local actions. The suitability of these maps for the questions that they supposed to throw light on must thus be verified to limit the risks of unsuitability and priority given

[1] UAA: Useful agricultural area

to a representation method matched to the questions posed. For example the compatibility of HNV maps with the assessment of very targeted or local actions such as agro-environmental measures (AEM) will need finer data (do not confuse zoning maps and sites map) (SOLAGRO, 2006).

As part of the **European RUBICODE project** (Rationalising Biodiversity Conservation in Dynamic Ecosystems), new conservation concepts are being developed, based on the capacity of ecosystems to experience a disturbance whilst maintaining their ecological functions and services (ecological resilience). This **dynamic ecosystem approach** is intended to complete the current strategic conservation approaches, generally developed around a more static vision of nature. Within this framework, research is underway to develop ecological quality monitoring indicators for ecosystems and habitats (Da Silva *et al.*, 2008). Considering that the ecological characteristics of plants and animals can be associated with specific ecosystem functions, they then constitute a promising functional biodiversity indicator. In addition, as these characteristics break down to different geographical scales they provide useful indications at both regional and bio-regional scales.

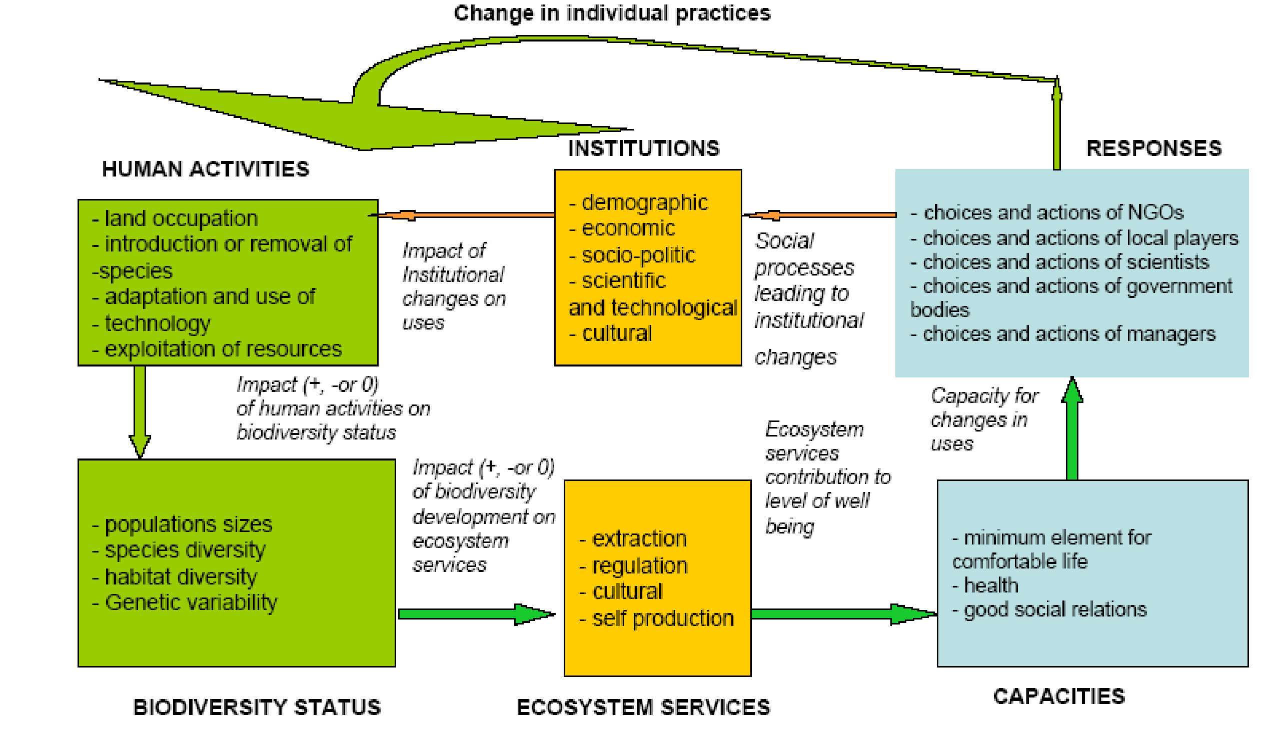
The major types of ecosystems considered are (Sousa *et al.*, 2008): fresh water, forests, semi-natural grasslands and shrublands.

The **pressure-state-response (PER) indicators**, intended to evaluate anthropogenic pressures on biodiversity and social responses and that can be used to compensate for the negative effects of pressures, have a central place among the indicators of society-nature interactions. Their analysis framework, a source of several ambiguities (ecological interactions and biodiversity adaptive responses not taken into account, different pressure or state classifications according to the players…), was reviewed by Levrel (2007) who proposes an alternative framework taking PSR indicators and the *Millennium Ecosystem Assessment* framework (figure IV-11) into account. This schema is a good pedagogical illustration of the working group's multidimensional analysis framework, from which economists and ecologists must obtain their responses.

Finally, we need to improve the kit of indicators in two respects:

- **define "risk indicators"** (or sensitivity or vulnerability indicators), for evaluating an entity's (a population, a species, an ecosystem) degree of sensitivity to disturbances. We are beginning to have such indicators for populations, through **population viability analyses**, which, using demographic and genetic data, can be used to establish a diagnosis of their risk of extinction,
- specify the **spatial validity domain for indicators**, meaning the possibility of using them to monitor the change in biodiversity at a given spatial scale.

Figure IV-11: Alternative framework for the identification of interaction indicators




*Source : Levrel, 2007*

# 4. Biodiversity and ecosystem services

## 4.1. The concept of ecosystem services

In chapter II we presented the concept of ecosystem services[1] and the type classification proposed by the MEA (*Millennium Ecosystem Assessment*) in four groups, that we present again in figure IV-12.

This typology in fact distinguishes two sets:

First, it identifies **"maintenance services"** not directly used by man but which condition the correct operation of the ecosystems, in the short term, but also in their ability to adapt in the long term: nutrient recycling ability, pedogenesis (formation of soils from the mother rock), the importance of primary production as the first link in food chains, resistance to invasion by foreign species, etc.

Next, arising from the maintenance services, it characterises the **services in the strict sense**, used by man and which the MEA proposes to divide into three groups:

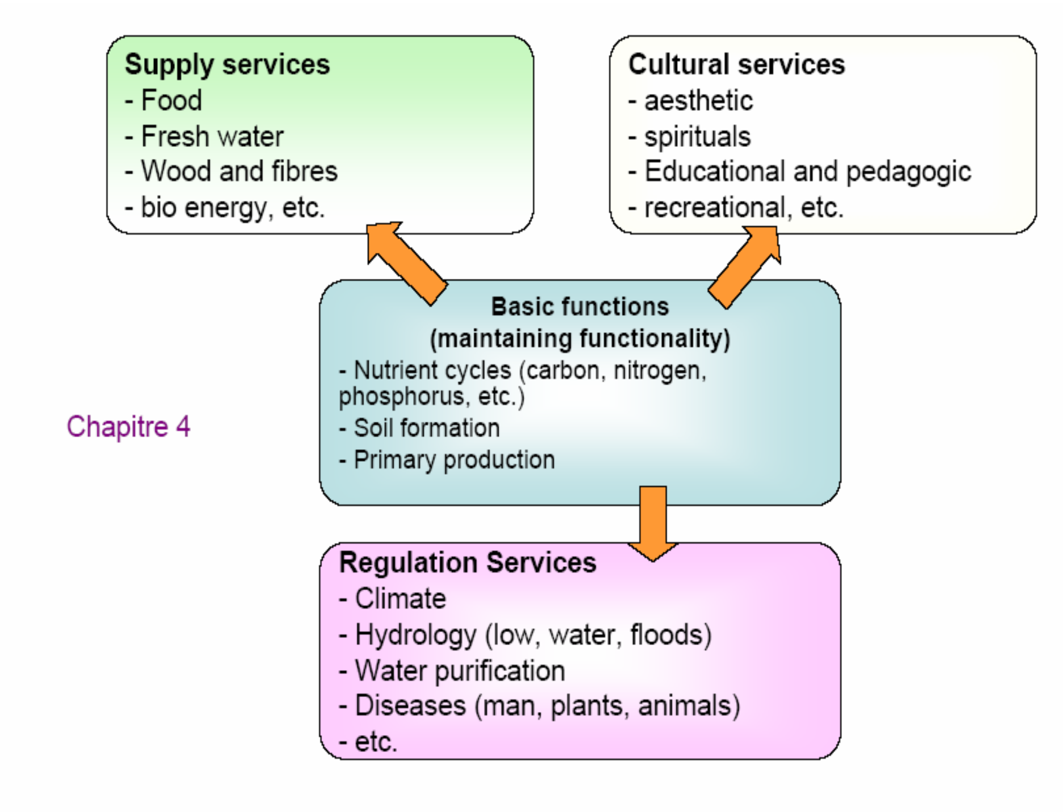
- the "supply services" (or extraction services) that lead to "appropriable" goods (foods, materials and fibres, fresh water, bio-energy), whether these goods are self consumed, bartered or marketed,

[1] People also talk about "services supplied by ecosystems" or "ecological services".

- the "regulation services", meaning the ability to modulate phenomena like climate, the occurrence and amplitude of diseases (human but also animal and plant) or different aspects of the water cycle (floods, minimum flows, physical and chemical quality) in a direction favourable to man, or to protect against catastrophic events (cyclones, tsunamis, torrential rain); contrary to supply services, these regulation services cannot generally be appropriated and have rather the status of public goods

- the "cultural services", meaning the use of ecosystems for recreational, aesthetic and spiritual (for example nature as a source of artistic creation or comfort) or educational purposes (for example see the green and blue classes, but also the role, evoked in Chapter II, that imitation of nature can play in innovation).

Figure IV-12: Ecosystem services classification according to the MEA




## 4.2. Function *versus* ecosystem service

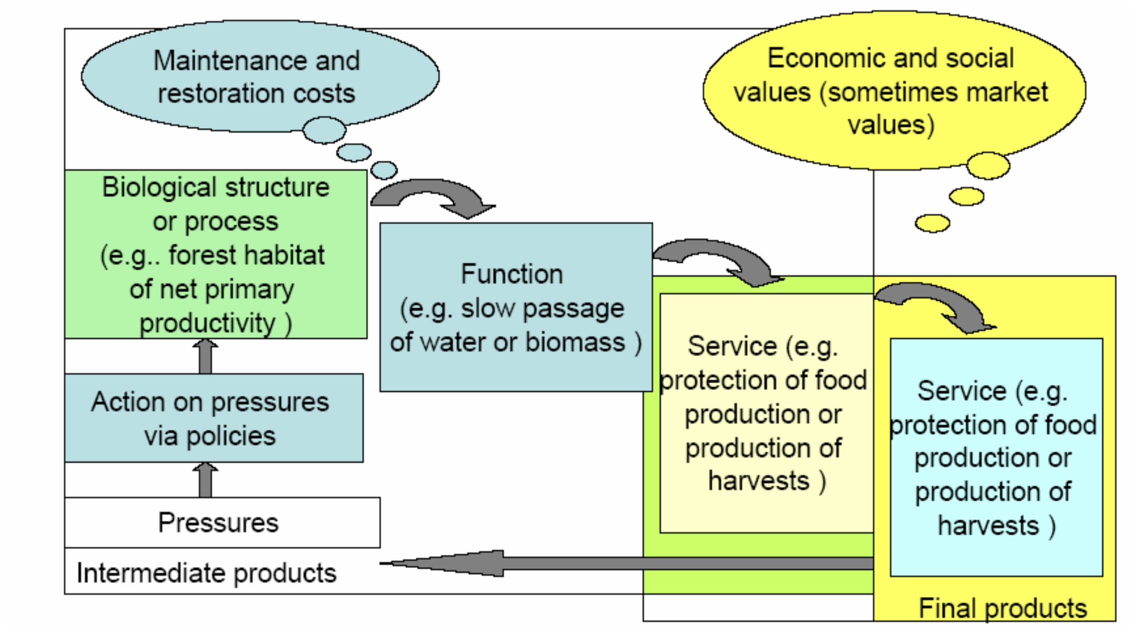
The ecological functions of an ecosystem, the processes of correct working of a system, must be distinguished from the ecological services it renders, which are the result of the correct working. We thus distinguish what ecosystems provide through their functions from the benefits that human beings draw from it. For example, the provision of foodstuffs (supply service in the international MEA) is not a service directly provided by the ecosystems, as the majority of food products are currently produced by agricultural activity. Ecosystems only supply supports to agriculture (cultivable areas, animal and plant species); only hunting, fishing and gathering products provide foodstuffs directly produced by the ecosystems.

Most of the benefits supplied by ecosystems are indirect. Complex ecological processes and changes undergone by ecosystems contribute to the production of these services (see figure IV-13, European Commission, 2008).

A group of experts from the AEE recently proposed a definition of ecosystem services and functions at the international workshop on the theme "Common international classification of ecosystem services" (December 2008): "ecosystem services are the products (*outputs*) of ecosystem functions that contribute directly to human well-being"

Taking these ecological functions into account can open a new field of possibilities in the choice of policies and associated tools aiming for a better conservation approach and the sustainable use of biological diversity. In fact, their valuation and accounting will enable us to preserve of the natural capital that is essential for the provision of ecological services whilst enabling us to respond to the needs of human societies (com. D. Couvet, MNHN).

**Figure IV-13: Link between biodiversity and production of services by ecosystems**




Source: European commission, 2008

*The ecosystem services indicators*

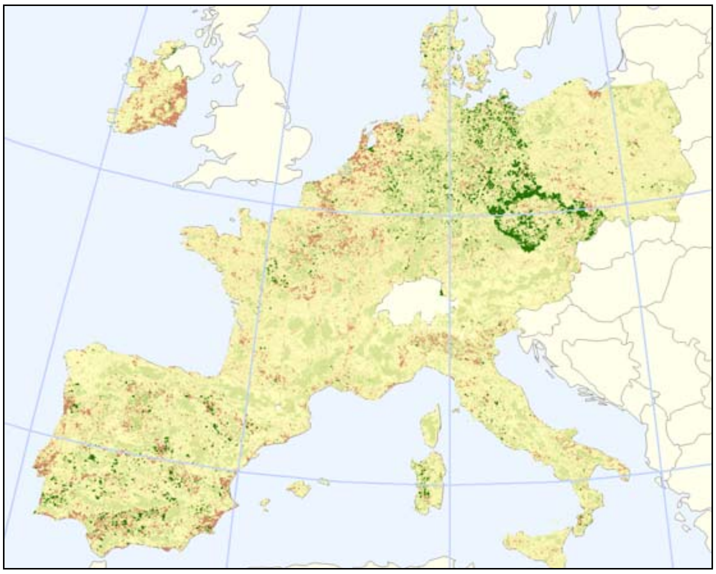
Various indicators can currently be used to measure the importance of these services:

- **Measurement of the correct working** of ecosystems can be based on various methods for monitoring the dynamics of major bio-geo-chemical cycles: for example we can measure the denitrification activity of water courses or the respiration of micro-organisms in the soils, indicators of their activity or the photosynthetic intensity of vegetation cover, which is related to primary productivity. Other measurements can be obtained by remote sensing. So the European Environment Agency (Weber, 2008) has constructed from such measurements, for the whole European territory and on a 1 km² grid, an indicator called the nLEP, for *net Landscape Ecological Potential*. This indicator, whose changes can be measured (figure IV-14) is the combination of four sets of geographical data:

- *The Green Background Landscape Index* (GBLI) which expresses a vegetation potential for the area, according to the land use intensity. The values are calculated from the "CORINE Land Cover" satellite database and benefit from successive updates.

**Figure IV-14: Net landscape ecological potential (nLEP) variation for Europe, 1990-2000**

(The green zones are those where the ecological potential has improved, those in red where it has been damaged)



*Source: Weber 2008*

- *The value attributed to nature,* evaluated using the importance of their designation by science and public authorities. This indicator is calculated from national, European (Natura 2000) and international databases. The indicator thus captures information that cannot be seen with satellite images, meaning the richness of the territory in terms of species and habitats.
- *Landscape fragmentation* by routes and railways, which is not captured in the two previous layers. The selected indicator is the "effective mesh size" (MEFF), which is the inverse of this fragmentation and conditions the space that can be explored by land animals.
- *The landscape heterogeneity,* which favours biodiversity by the multiplication of niches and transition zones The calculation of this segmentation indicator by satellite images is known but has not yet been integrated into the nLEP.

**However, these remote sensing approaches should be validated by measurements on the ground to judge their ability to monitor changes in biodiversity.**

For several years, the European Environment agency has been engaged in a methodological review to define an accounting framework for European ecosystems. Based on the satellite data provided by CORINE Land Cover, combined with ground data, we can thus evaluate the loss of certain ecosystem types over time under the effect of certain pressures, notably urbanisation and intensive agriculture, and deduce the loss of services potential as approximated by the nLEP indicator. The methodology has been tested first on one ecosystem type, the Mediterranean coastal wetlands, and validated by comparison with ground data from the Camargue (FR), Do•ana (GR), Danube delta (RO) and Amvrakikos (GR) sites.

The work is currently continuing as part of a methodological contribution to the TEEB process, 2nd phase, to analyse the transition from physical accounting for these coastal wetland area ecosystems to monetary accounting. In addition, the extension of the analysis to other ecosystem types is envisaged (EEA, 2008).

- **Supply services** can be monitored directly using the quantities of goods from these ecosystems. This estimate is easy for products put on the market, but more difficult for self consumed or bartered products, which can sometimes represent substantial part. So, it is estimated that 40% of the wood collected in French forests is self consumed (FPF, 2008). Similarly, Coates (2002) estimates that, for set of eight countries in South East Asia, the official FAO figure for fisheries in inland waters must be multiplied by at least three to take account of self consumption and obtain an estimate close to reality. This is also what the strengthening during the last few years of the inspection of fish landing by the DOM (French Overseas Departments) veterinary services reveals. In addition, we should distinguish the concepts of "sustainable potential service" – meaning quantities that can be taken from eco-systems without compromising their production capacity – and "actually used service". Depending on the case, the service can be used very partially (for example the supply of freshwater in Scandinavia) or on the contrary excessively (the case of over exploitation of numerous fish stocks). Direct indicators, like the extraction rate (estimated, for example, at 69% for French forests in 1998-2002; MAP, 2006) or indirect, like the Marine Trophic Index (see III.3) must thus complete the measurement of quantities taken.

- **Regulation services** can be approached more or less directly. For example, fairly good models are available for the prediction of hydrographs (river flow rate throughout the year) from the rainfall measurements, morphology and vegetation coverage type in the catchment area. For climate, most works has concentrated on the role of ecosystems in $CO_2$ sequestration or release, due to its contribution to the greenhouse effect. The data on other greenhouse gases (methane, nitrogen or sulphur oxides) is more fragmented and limited to particular ecosystems (wetlands, intensive farming). These approaches are thus centred on the modulation of the global climate of the planet. In contrast, the effect of ecosystems on the local climate is, as we will see, less documented and, in addition, controversial. Finally, as far as "biological moderation" services (reduction in diseases, limiting the proliferation of invasive species, etc) are concerned, the approaches remain fairly qualitative and above all based on the *a priori* favourable effect of spatial heterogeneity and landscape mosaic (juxtaposition of different ecosystems at varied scales).

- Finally, **cultural services** are most often approached through the numbers visiting ecosystems, in different forms (tourism, hunting and fishing, nature sports, educational visits). These estimates can be completed, as we will see, by measurement of the interest of these ecosystems for people who do not physically visit them but for whom the existence of diverse complex nature, escaping from human control is a source of scientific, artistic, philosophical or religious inspiration (for example, the "Earth Keeping" movement).

## 4.3. What are the relationships between biodiversity and ecosystem services?

Having defined the main concepts and indicators linked to biodiversity on the one hand, and ecosystem services on the other, we should now discuss the link between theses two concepts and to say what will be likely to be taken into account by the economic analysis.

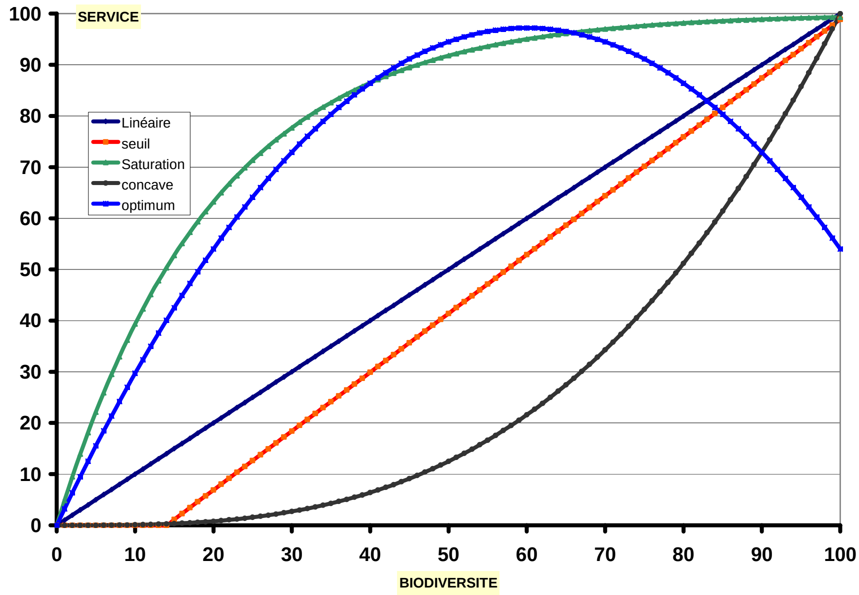
**This question of the relationship between biodiversity and ecosystem services is rarely explicitly posed**, in so much as it seems obvious that these services – being completely the product of living activity – can only suffer very directly from a reduction in the abundance and diversity of these living beings. The two concepts often being cited in association, some could even ask if they are not if fact two formulations designating two facets of the same thing, as when "nature and landscape" are evoked.

But though there is no doubt about this "existential' and qualitative link, the question of the "form" and intensity of this relationship is the subject of numerous discussions and an abundant scientific literature1. This question that may appear to be purely academic is in fact crucial for our work. In fact:

- As we have indicated, biodiversity is only currently known in a very partial and doubtless biased way, and the capacity of the currently available indicators to account for changes in the whole of biodiversity is still very imperfect. This means that certain biodiversity variations will only be detected indirectly, via any variations in ecosystem services to which this biodiversity contributes.
- As we will see later, the currently most numerous and the most successfully complete economic works have related much more to the evaluation of ecosystem services than to biodiversity. Accepting that an economic assessment of these services and their fluctuations should be considered as a measure of the value of underlying biodiversity variations assumes that a link between these two aspects is clearly established.

[1] We will only cite a few articles giving a glimpse of these discussions and providing a more detailed bibliography: Johnson *et al.* (1996), Wardle *et al.* (1997), Loreau and Behera (1999), Chapin III *et al.* (2000), Loreau *et al.* (2001), Hooper *et al.* (2005), Worm *et al.* (2006), Danovaro and Pusceddu (2007), Danovaro *et al.* (2008), Lanta and Leps (2008).

The available scientific works have been interested in biodiversity essentially in its species related dimension, describing it in terms of systematic (number of species) or functional (by grouping together species having a similar economic role) entities. Based on both theoretical approaches and observations on the ground, they consider extremely varied possibilities (figure IV-15).




*Source: B. Chevassus-au-Louis*

So, even the idea of a "positive monotonic" relationship, meaning a progressive increase in ecological services with increase in biodiversity is not totally admissible and some authors defend the possibility of more complex relationships, with links that can sometimes be positive, sometimes negative (model "with optimum" in the figure).

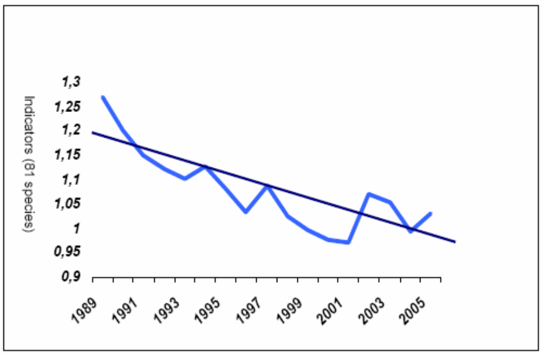
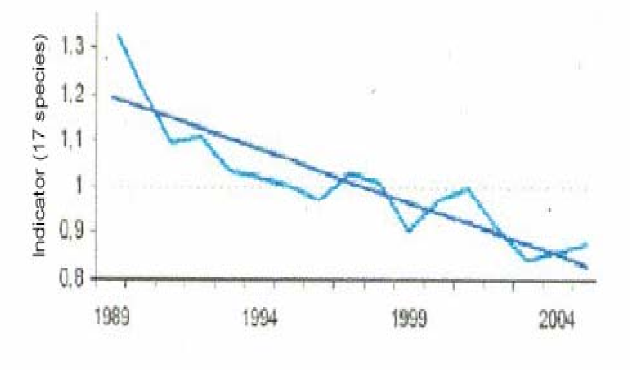
As far as the intensity of this link is concerned, Naidoo *et al.* (2008) emphasise, for example, that at planetary level there is no spatial coincidence between regions with high species biodiversity and those supplying important ecosystem services (the services considered being carbon storage, the supply of fresh water and the production of livestock fodder. The origin of this discordance is doubtless linked, at least partly, to the fact that biodiversity is estimated in this study solely by the species diversity of well inventoried major groups (vertebrates, higher plants), which, as we have said, only represents one of the facets of biodiversity.

Even among the most currently accepted "positive monotonic" models, three extreme cases can be distinguished.

**The simplest case is that of a linear relation, in which proportionality between the variations of two factors is assumed.** To take an example, three ecosystem services indicators can be derived it is possible from the abundance index for common birds (figure IV-16),  linked in a linear way to this abundance index:

- An indicator of plant dissemination capacity, which can be a component of ecosystem maintenance services; this index is based on the abundance of seed eating birds.
- An indicator of crop pests (supply function) based on the abundance of insectivorous birds.
- A "cultural service" indicator, based on the abundance of the 14 common bird species cited in the *Fables* of La Fontaine and which measures the erosion of this "identity heritage".

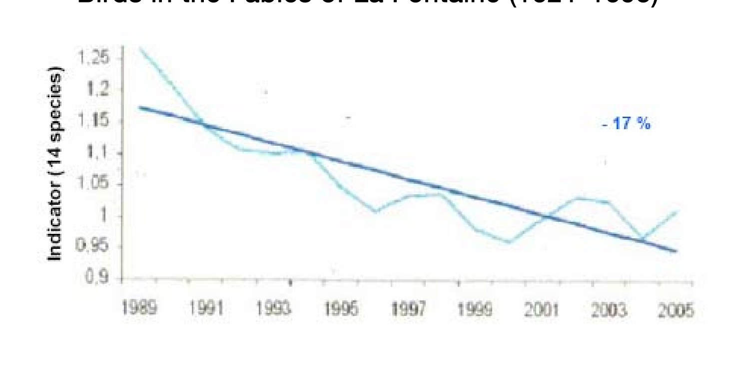
**Figure IV-16: Examples of ecosystem service indicators that can be calculated from a biodiversity indicator, the STOC index of common bird abundance**

| Dissemination indicator | Insect eating species in agricultural environments |
|---|---|
|  |  |





Birds in the Fables of La Fontaine (1621-1695)




*Source: IFEN, Jiguet (unpublished)*

**The "convex" model ("saturation" model of the figure) is based on the idea of a certain "functional redundancy" of species** (several species playing the same ecological role). It supposes that a notable reduction in biodiversity will only be translated, at least in the beginning, by relatively limited reductions of ecological services. This type of relation can particularly exist when there are saturation phenomena: so, when a population of pollinating insects is abundant and diversified, the fertilisation of entomophilous plants is fully achieved and an increase in the population of pollinators will have no effect on this service.

**In contrast, the "concave" model brings out the major role of certain "key" species whose disappearance can have major consequences.** On the contrary, it predicts that variations, even small, of biodiversity could translate into a high reduction in ecosystem services. Even in the absence of such key species, if ecosystem services depend, as we have emphasised, not only on the presence of species but above all on the interactions between them, this number varies as a function of type $n^n$, where $n$ is the number of species. It will thus decrease very strongly with reduction in the number of species.

**Finally, we can have "threshold" 'models**, linear or not, in which the service varies sharply, even ceases, below a certain non zero biodiversity value.

## 5. Options for the economic approach

To take these analyses into account and link them to the economic analysis, the working group has decided to use the four options below:

**First option, to distinguish in the biodiversity of a given territory a "heritage" dimension**, meaning the existence of entities designated as such in official inventories, possibly protected, or for which such procedures have been initiated, **and its "general" or "functional" dimension,** meaning the presence of more or less abundant ordinary entities contributing to the production of ecosystem services.

This distinction can concern the organisation of the different levels of biodiversity:

- At the genetic level, we can have a local domestic or wild population presenting original characteristics but must also consider the "ordinary" genetic diversity between individuals, which, without any individual presenting any particular interest, will allow the general evolution of the species.
- At species level, we may have "emblematic" species registered on protection lists, and the procession of ordinary species, which contribute to the supply or ecosystem services through their interactions.
- At the ecological level, we could identify in an area, on the one hand, particularly original habitats, rare on the regional or national scale, or remarkable landscapes (classified sites) and, on the other, a mosaic of ordinary habitats making up the essential part of the landscapes and responsible for their functional characteristics.

**Of course, we could discuss the restrictive and contingent character of this criterion assimilating "noticeable biodiversity" and "noted biodiversity", meaning effectively taken into account by various protection initiatives**, even giving a very wide meaning to this concept (for example it can include the existence of an awareness campaign or the creation of a protection association). In fact, it is obvious that, even accepting the diverse "eligibility" criteria – ecological assessment but also common knowledge, cultural criteria –, this "noted" biodiversity will be conditioned by the available knowledge and will not include many items of information of major interest that could be revealed in the future. To take an example, the dodo on the island of Mauritius, that great flightless bird that disappeared in the middle of the XVIII$^{th}$ century, now makes poultry selectors dream, thinking that it could have made a much superior farmed animal to our current turkey. Similarly we indicated at the end of chapter II how the nanostructures of living things, still little described, could in future constitute a major source of inspiration for nanotechnologies.

**For this reason, even within the limits of utilitarian ethics, any biological entity can present a potential value whose loss, as soon as it is irreversible, indeed irremediable (in the sense that it is judged impossible to compensate for it), must have an infinite, or rather, inestimable cost[1]. In opposition to this theoretical reminder, the economic analysis also emphasises that these "unnoticed" entities, of which many disappear every year[2], have *de facto* a zero or quasi zero value for today's society, if we attempt to measure this implicit value by various methods.**

It is thus to escape from this dilemma that the group has suggested applying a dichotomy between "noticed" biodiversity and general (or "ordinary") biodiversity and treating these two groups in a different manner. Whilst proposing this principle and some guidelines, the group did not wished to specify further the designation criteria that could be chosen to identify these noticeable elements of biodiversity. In fact it appears legitimate, and conforms to reality, that this designation procedure should mobilise, depending on the environment, the players and entities concerned, criteria that may be different and according to a hierarchical ordering that is difficult to do in advance.

Whilst we are aware of the limits of the analogy, this approach can be considered to be fairly similar to the one that distinguishes "historic" monuments and others, the first benefiting from protection and/or support measures from society, for which we can measure the concrete form – sometimes limited, it can be added – by estimating the financial resources used.

**The second option is to asset straight away that "noticeable" biodiversity must be considered taking into account multiple ecological, ethical, cultural, aesthetic criteria, having presided in its identification, and as a result, the economic analysis can only play a subsidiary and indicative role in this case (as in the case of historic monuments).**

To us this option appears justified for several reasons:

- The first is that the "heritage" elements have, often, become rare – this is also what has lead to their identification – which obliges us to recognise that their contribution to the supply of ecosystem services is undoubtedly, alas, limited; it is thus irrelevant to evaluate them via any variation of these services.
- The second reason is that an economic evaluation **only has meaning – other than academic – if we agree to enter into a logic of exchange and "potential substitutability"** , in other words if we admit, even with certain precautions, that the entities evaluated could be exchanged against other entities judged to be equivalent, this equivalence being estimated on the basis of the value in monetary or possibly other

[1] As we will see, economic theory recommends a zero discount rate for these non-substitutable goods.
[2] Even with the moderate assumptions of a current extinction rate of 0.01% per year and ten million species (limiting ourselves to "organised|" beings, formed of an assembly of eukaryotic cells, meaning having a true core), about 1,000 species disappear every year.

accounting units. If, on the contrary, we have decided that these entities must be protected, that they are part of an inalienable natural heritage, this decision, even if it could be reconsidered, takes its place in the logic of a "political" action, similar to that for cultural heritage, which transcends economic rationality. However, we note that, even when protected, these entities could be subject to unintentional, or intentional damage, and that in this case an *ex-post* monetary assessment of the loss could be useful.

- Finally, on a more pragmatic level, the economic values that could be associated with these heritage elements are often "non-use" values (option, existence or legacy value) for which economic evaluation methods appear, as we will see, particularly fragile.

**Third option, the group proposes to address the economic analysis of general biodiversity not directly but through the ecosystem services to which it contributes.**

This option is consistent with the accent that the working group has placed, in the definition of biodiversity, on the importance of multiple interactions between its components – individuals, species, ecosystems – but also on the very largely unknown character of a large part of these components. It is thus theoretically preferable but also more realistic to assess the global resultant of these multiple interactions within the system comprising the biodiversity rather than to try to identify the contribution belonging to each component.

Here too, the simplistic character of this option must not be masked. It results, in particular, from two things:

- the analysis of ecosystem services mainly includes use values, which we will see only comprise part of the total value, even when enlarged to future or unconditional uses,
- from the viewpoint that we have chosen, general biodiversity "contains" "unnoticed" entities with some likely to be noticed in the future. However, these entities will only be accounted for through their contribution to these ecosystem services.

**Finally, as a fourth option, the group proposes to work on the "median" hypothesis of a positive linear relationship between general biodiversity and ecosystem services.**

This median hypothesis, which could be debated, is based on the fact that the "convex" models, that reduce the potential effect of a reduction in biodiversity, often consider the "instantaneous characteristics" of ecosystems and rarely longer term characteristics when faced by varied perturbations (resistance, resilience, adaptability) that could depend on apparently "optional" entities at a given moment. Conversely, the few empirical observations in favour of concave models do not drastically deviate from a linear type relation.

**In addition, we have previously insisted on the necessity of including the idea of absolute abundance, in particular the size of populations, in the biodiversity concepts and indicators and not limiting ourselves to diversity measures in the strict sense.** From this viewpoint, a reduction in the size of populations will be considered as a loss of biodiversity not solely because it could eventually lead to the disappearance of the species, but because it leads immediately to a possible reduction of the role of this species in the working of the ecosystem. We also add that this point of view is completely justified in the case of genetic diversity, which is effectively reduced when the size of a population diminishes. From this viewpoint, the relation between biodiversity and ecosystem services appears fairly naturally, if not linear, at least monotonically increasing.

This option leads us to consider that the economic evaluation of the reduction in services will provide an acceptable measure of the associated losses of biodiversity. This median hypothesis also allows us to consider that observed or potential variations in biodiversity in a given environment could be "monetarised" assuming a similar relative variation of the ecosystem services of these environments: for example, if we take the STOC indicator of the abundance of common birds, a 30% fall in this indicator in agricultural environments would be evaluated at 30% of the value of the agro system services considered.

**These four options thus mean that:**

- **In ecosystems without noticeable biodiversity (meaning "noticed" as a given instant), we could apply the economic evaluation of ecosystem services as reference, in a substitutability logic.**
- **In ecosystems including such noticeable biodiversity elements, the economic evaluation will only be useable and, *a fortiori*, used if a prior examination of the value of these elements, emphasising other than economic criteria, will have actually allowed the conclusion that it was acceptable to enter a substitutability logic, after satisfactory solutions had been found for these noticeable biodiversity elements.**

## Conclusion

This chapter has emphasised the multiple dimensions of the biodiversity concept, and in consequence, the need to have recourse to a large palette of indicators to define it. It has also emphasised how important it is to go beyond a vision based on single easily observable known species, in particular if we want to understand the contribution of biodiversity to the supply of ecosystem services.

This complexity could lead to the conclusion that we cannot currently possible to make a general judgement, even qualitatively, on the state of biodiversity in a given ecosystem. Using the human health metaphor again, we could say that such a pessimistic conclusion has as little relevance as that which would see the multiplication of biomedical analysis tools as an obstacle to judging the good health of an individual.

**Conversely, let us accept that ecological evaluation is now likely, with the collection of the necessary data, to provide a diagnosis of the state and future of biodiversity and ecosystem services in a given location when faced with a possible modification of the environment.**

The second part of this analysis has shown the way in which the concept of ecosystem services could be used to address the economic evaluation of "ordinary" biodiversity, with a debatable but reasoned hypothesis of coupling between the extent of biodiversity and abundant provision of these services.

**Finally, without prohibiting the economic approach from demonstrating its possible input to the analysis of noticeable biodiversity, we have emphasised that it was doubtless preferable to mobilise other values to treat the relevant management of this heritage.**

## Bibliographic References

Aubertin C. (2000), « L'ascension fulgurante d'un concept flou » ("The Dazzling Rise of a Fuzzy Concept"), *La Recherche (Research* Magazine*)*, n° 333, p. 84-87.

Barbault R. et Chevassus-au-Louis B. (2004), *Biodiversity and Global Change*, ministère des Affaires étrangères (Foreign Affairs Ministry), ADPF, 237 p.

Biala K., Terres J. M., Pointereau P., Jiguet F., Bas Y., Paracchini M. L. et Calvo S. (à paraître), *Identification of High Nature Value Farmland Based on Statistical Information and Farm Practices Surveys: A Case Study of France*.

Biggs R., Scholes R. J. et Reyers B. (2004), *Assessing Biodiversity Intactness at Multiple Scales*, Bridging Scales and Epistemologies Conference, Alexandria, Egypt.

Butchard S. H. M., Stattersfield A. J., Bennum L. A. (2004), « Measuring global trends in the status of biodiversity: red list indices for birds ». *PLoS Biology*, 2, 1-11.

Chapin III F. S., Zavaleta E. S. *et al.* (2000), « Consequences of changing biodiversity », *Nature*, 405, 234-242.

Chevassus-au-Louis B. (2007), « La biodiversité: un nouveau regard sur la diversité du vivant. I. Immensité et complexité » (Biodiversity: a New Look at Living Species I. Immenseness and Complexity), *Cahiers Agricultures (Agriculture Journals)*, n° 16, p. 219-228.

Chevassus-au-Louis B. (2008a), « La biodiversité: un nouveau regard sur la diversité du vivant. II. Stabilité et utilité »,(Biodiversity: a New Look at Living Species II. Stability and Usefulness) *Cahiers Agricultures*, *(Agriculture Journals)* n° 17, p. 51-57.

Chevassus-au-Louis B. (2008b), « La biodiversité: un nouveau regard sur la diversité du vivant. III. Fragilité: vers la sixième extinction (Biodiversity: a New Look at Living Species. III. Fragility: towards Extinction #6) *Cahiers Agricultures (Agriculture Journal)s*, 17, 303-313.

Coates D. (2002), *Inland Capture Fishery Statistics of Southeast Asia: Current Status and Information Needs*, FAO /RAP n° 2002/11, 121 p. (site: www.fao.org/documents/)

Commission européenne (European Commission) (2008), *L'économie des écosystèmes et de la biodiversité (Economy of Ecosystems and of Biodiversity)*, Luxembourg, Office des publications officielles des Communautés européennes (Office in Charge of Official Publications for European Communities), 64 p.

Couvet D., Teyssèdre A., Jiguet F., Julliard R. et Levrel H. (à paraître) (to be Issued), *Indicateurs et observatoires de biodiversité (Biodiversity Indicators and Watch Bodies)*.

Couvet D., Jiguet F., Julliard R. et Levrel H. (2007), « Observatoires et indicateurs de biodiversité », (*Biodiversity Watch Bodies and Indicators) Working Paper*. ( *in* Barbault R. et Chevassus-au-Louis B. (éds.), *Biodiversité et changement globaux*, ministère des Affaires étrangères (Foreign Affairs Ministry), ADPF, 241 p.

Da Silva P. M., Sousa J. P., Feld C. K. et Hering D., *Web Report Comparing Indicators Targeting Habitat Area*, 34 p., www.rubicode.net/rubicode/RUBICODE_Report_on_Indicators_of_Habitat_Area.pdf.

Danovaro R., Gambi C. *et al.* (2008), « Exponential decline of deep-sea ecosystem functioning linked to benthic biodiversity loss », *Current Biology*, 18, 1-8.

Danovaro R. et Pusceddu A. (2007), « Biodiversity and ecosystem functioning in coastal lagoons: Does microbial diversity play any role? », *Estuarine, Coastal and Shelf Science*, 75, 4-12.

Devictor V., Julliard R., Clavel J., Jiguet F., Lee A. et Couvet D. (2008), « Functional biotic homogenization of bird communities in disturbed landscapes », *Global Ecology and Biogeography,* 17, 252–261.

Devictor V., Julliard R., Couvet D. et Jiguet F. (2008), « Birds are tracking climate warming, but not fast enough », *Proc. R. Soc. B* (doi:10.1098/rspb.2008.0878) Published online.

Donald P. F., Green R. E. et Heath M. F. (2001), « Agricultural intensification and the collapse of Europe's farmland bird populations », *Proc. Roy. Soc. London Series B*, 26, 25-29.

European Environment Agency (2008), « Ecosystem services, accounting for what matters », *EEA Briefing 2008-02*, Office for Official Publications of the European Communities.

European Environment Agency (2007), « Halting the loss of biodiversity by 2010: proposal for a first set of indicators to monitor progress in Europe », *EEA Technical Report*, 11, 182 p.

Hooper D. U., Chapin III F. S. *et al.* (2005), « Effects of biodiversity on ecosystem functioning: a consensus of current knowledge », *Ecological Monographs* 75, 3-35.

Hubbell S. P. (2001), *The Unified Neutral Theory of Biodiversity and Biogeography*, Ed. Princeton University Press, Princeton (New Jersey) 375 p.

IFB (Institut français de la biodiversité) (French Institute for Biodiversity) (2008), Commission scientifique de l'IFB (IFB Scientific Commission), *Réflexion stratégique*, *Bilan des groupes de travail,* (Strategic Reflections, Work Group Achievements) O80 p.

IFEN (Institut français de l'environnement (French Institute for Environment) (2003), « 45 indicateurs de développement durable: une contribution de l'IFEN »("45 Sustainable Environment Indicators, an IEFN Contribution"), *Études et Travaux (Studies and Works)*, n° 41, 110 p.

Johnson K. H., Vogt K. A. *et al.* (1996), « Biodiversity and the productivity and stability of ecosystems », *Trends in Ecology & Evolution*, 11, 372-377.

Julliard R., Clavel J., Devictor V., Jiguet F., Couvet D. (2006), « Spatial segregation of specialists and generalists in bird communities », *Ecology Letters*, vol . 9, n° 11, 1237-1244(8).

Lanta V., Leps J. (2008), « Effect of plant richness on invisibility of experimental plant communities », *Plant Ecology*, 198, 253-263.

Levrel H. (2007), « Quels indicateurs pour la gestion de la biodiversité ? » (Whch Indicators for Biodiversity Management), *Cahiers de l'IFB (IFB* , 95 p.

Loreau M., Naem S. *et al.* (2001), « Biodiversity and ecosystem functioning: current knowledge and future challenges », *Science*, 294, 804-808.

Loreau M. et Behera N. (1999), « Phenotypic diversity and stability of ecosystem processes », *Theoretical Population Biology*, 56, 29-47.

MAP (2006), *Les indicateurs de gestion durable des forêts françaises*, (Indicators for a Sustainable Managemrent of the French Forests) édition 2005, 148 p.

MEEDDAT (2007), Présentation des indicateurs de suivi de la biodiversité proposés pour la métropole, (Presentation of the Biodiversity Follow-Through Indicators Proposed for Metropolitan Territory) *rapport MEEDDAT/DNP/Cellule biodiversité*, 63 p.

Naidoo R. et Balmford A. (2008), « Global mapping of ecosystem services and conservation priorities », *PNAS,* 105, 9495-9500.

Organisation de coopération et de développement économique (OCDE) (OECD) (1994), *Indicateurs d'environnement: Corps central de l'OCDE*, Paris, OCDE.

Pauly D. et Watson R. (2005), « Background and interpretation of the 'Marine Trophic Index' as a measure of biodiversity. Philosophical Transactions of the Royal Society », *Biological Sciences*, n° 360, p. 415-423.

Pauly D., Christensen V., Dalsgaard J., Froese R. et Torres F. (1998), « Fishing down marine food webs », *Science*, 6, vol. 279, 860-862.

Scholes R. J. et Biggs R. (2005), « A biodiversity intactness index », *Nature*, vol. 434, 45-49.

SOLAGRO (2006), *Cartographie des zones agricoles à haute valeur naturelle*, (Mapping of Agricultural Zones With a High Natural Value) rapport Commission européenne-SOLAGRO, 96 p.

Wardle D. A., Bonner K. I., Nicholson K. S. (1997), « Biodiversity and plant litter: Experimental evidence does not support the view that enhanced species richness improves ecosystem function », *Oikos*, 79, 247-258.

Weber J.-L., Uhel, R., Spyropoulou R., Breton F., Arévalo J., Richard D., Haines-Young R., Potschin M., Kumar P., Martin B., Lomas P., Gomez E., Tomas P., Ezzine D., Nichersu J. et Marin E.

(2008), Draft report on *Ecosystem Accounting for the Cost of Biodiversity Losses: Framework and Case Study for Coastal Mediterranean Wetlands*.

Weber J. L. (2008), « Comptabilité des écosystèmes et de leurs services », (Compatibility of Ecosystems with their Services) *12e colloque de l'Association de Comptabilité nationale*, Paris, 4-6 juin 2008.

Worm B., Barbier E. B. *et al.* (2006), « Impacts of biodiversity loss on ocean ecosystem services », *Science*, 314, 787-790.

# Chapter V

# The economic evaluation of biodiversity and ecosystem services: the state of scientific knowledge

There are two basic reasons that have led people to consider that biodiversity and ecosystem services are relevant subjects for economic analysis. The first is that biodiversity is a source of value in society. For many reasons that have already been mentioned and that we will go into later, greater biological diversity generally results in greater well-being. The second is that human choices and behaviours in society have and will continue to have undeniable impacts on biodiversity. We have seen that these choices have already led, often in an unintentional way, to a decrease in diversity, whatever the method of measuring it, and will continue to do so. Biodiversity and ecosystem services seem to be rare and useful resources and are therefore suitable for investigation using the conceptual and methodological framework of economics.

However, many services provided by ecosystems have no price because, as there are no clearly defined property rights, they cannot be the subject of transactions on the market. Nevertheless, they contribute to the well-being of economic agents who may be sensitive to variations in their quality or the intensity with which they benefit from them, but may not necessarily be so, and we will see that this can be a problem. Noting this fact, economists have tried to measure the variations in utility caused by an increase or decrease in the quality of the environment and, more often than not, to do it in monetary terms so as to obtain values commensurate with the exchanges that take place on the market. This has in fact been a classical problem in economics ever since the engineer Jules Dupuit proposed the measurement of the consumer's surplus as the correct approximation of the social value of a bridge.

The question of economic evaluation is *priori* legitimate in a case where a decision maker must make a choice whose consequences affect other agents. This is particularly the case for public investment or private investments subject to public control due to their effect on third parties. This leads us to make a detour with the aim of explaining what is meant by economic evaluation and its relations with monetarisation (V.1.), before specifying the conceptual framework that can be used for the economic evaluation of biodiversity and ecosystem services (V.2.), presenting the main methodological problems raised by implementing it (V.3.) and then the main results provided by the economic literature on the essential aspects of this question. This chapter concludes with a description of the contributions of several recent initiatives in the field at both national and international levels (V.5.).

# 1. What do we mean by evaluation: what does economic evaluation mean

To evaluate is to determine a magnitude and give it a value. Economic evaluation is generally found in the context of a *cost/benefits analysis* (CBA). The principle of this is to compare the options involved in a choice and assigned to each of them the advantages and disadvantages that condition its social value. When this procedure is applied to issues that are easy to quantify and have no evident implications in the field of ethics, the use of a CBA does not run into too many difficulties or objections. Conversely, when the choices involve values considered to be ethical or intrinsic, the idea of calculations seems to be inappropriate and citizens would like to be able to rely on some fundamental principles to guide the choices that affect the community (for example, see Sagoff, 2004).

Thus, there seems to be a certain incompatibility between the ethical values that guide the right choices and the economic value, seen as a simple extension of market prices. Yet there is a certain continuity between these values. Obviously, the fact that there is a relationship excludes neither approximations nor errors nor any manipulations. But over a beyond these difficulties and imperfections, or even the real problems raised by this framework for analysis, the following pages should be read bearing in mind the question of whether it gives useful results for the objective that we have set ourselves and that means we have available.

## 1.1. Economic evaluation and ethical values

> "*Almost all economists are intellectually committed to the idea that the things people want can be valued in dollars and cents. If this is true, and things such as clean air, stable sea levels, tropical forests, and species diversity can be valued this way, then environmental issues submit – or so it is argued – quite readily to the discipline of economic analysis… Most environmentalists not only disagree with this idea, they find it morally deplorable*."
>
> (*The Economist*, 31st of January 2002)

The idea of value[1] is present in many fields of philosophical and social thought therefore reminding ourselves of the sense given to it by economists give is not otiose. The values to which citizens and public decision makers can refer when they justify their choices can belong to different orders of justification[2], but they entail costs that must be taken into account in a context of rarity and the alternative use of resources. We cannot presume to summarise the history of the theories of value here but we must resume some of the stages in their relationships with the question of the social justification of choices that involve all or part of society and which result in the theories of social choice.

---

[1] Webster's dictionary defines value as being the quality of something according to which we judge whether it is more or less desirable, useful, worthy or important.

[2] Several authors have proposed clarifications on the question of justification, in the field of the environment, for example, see Godard (2004).

## *a. The economic notion of value*

The history of economic thought shows that, even though it is limited *a priori* to an instrumental sense, the notion of value has taken on quite varied contents and forms before arriving at the present concept. It underlies mediaeval thought about "the just price" and was developed in quite contrasting ways by the mercantilists and the physiocrats, who considered work on the land to be the sole source of wealth, the aim of human work being to transfer to transform it in order to "bring out its value". Already, in the 16th century, Galiani defined value as a relation of subjective equivalence between goods and noted that it depended on usefulness and rarity. Starting from the end of the 18th century, A. Smith and the so-called "classical" economists made the distinction between use value and exchange value which reflects the cost of production by adding together the remuneration of labour, capital and land rent. Human work became the currency and the value of natural resources was measured by the work entailed in making them available and usable[1].

The origin of the current concept of the economic value can be found in the utilitarian philosophy of J. Bentham at the end of the 18th century. The starting point of his theory is that "ethical good" is an observable and demonstrable reality that can be defined on the basis of the elementary motivations of human nature: its "natural" propensity to seek happiness, i.e. the maximum pleasure and the minimum pain. Therefore, he proposed that individual and public behaviour be judged on the basis of their contribution to "greatest good of the greatest number", i.e. their social utility[2].

Utilitarianism can be characterised by a set of principles: "good" is defined as being well-being, actions are judged on the basis of their consequences and not the moral motivations of agents (consequentialism), the value of an action is a net balance of well-being (= pleasure – pain) independently of its distribution (therefore, a minority can be sacrificed) and individuals are interchangeable (impartiality and universalism). This means that doing good consists of maximising the sum of pleasure. An important feature of utilitarianism is in fact its "rationalism". The value of an act is "calculated" and not determined on the basis of principles having an intrinsic value. Therefore, this "arithmetic of pleasure", based on a sum of the consequences of an act on the well-being of all, assumes that those consequences can be measured and that their impact on the well-being of individuals can be assessed. John Stuart Mill introduced this philosophy more explicitly into economic analysis in the form of an indirect utilitarianism in which pleasure is only a means of arriving at the well-being[3] ("*welfare*") of the greatest number.

Obviously, taking people's well-being or happiness as the criteria for justifying collective choices is not the only possibility. We might want to maximise the country's

---

[1] But D. Ricardo gave them a value once again in his theory of differential rent: the value of land expresses its propensity to enhance the value of the work and capital of the people cultivating it.
[2] Just like its successors, Benthamite utilitarianism aimed to provide answers to a set of problems that confront societies: which principles guide behaviour of individuals, what are the tasks of government, how can the individual interests be reconciled with each other and how can individual interests be brought into agreement with those of the community.
[3] In economic science, the notion of well-being is not different from that of the social utility. It is not imply *a priori* any value judgements (even though Mill introduced one in the quality of pleasure) and should not be confused with the common idea of well-being as comfort items as opposed to essential needs.

power, promote the creation of society according to the Prince's taste, make society conform to a certain ideology or not limit the issues for choices to human interests alone … and we might want to make people happy in spite of themselves, like when elites consider that they are more able to make relevant choices in the interests of the populace because they are better educated and informed (we will come back to this point).

At the end of the 19th century, the "neo-classicists" (Jevons, Menger, Walras and Marshall) transformed the question of the maximisation of utility with the marginalist approach: this is the utility contributed by the unit gained or lost – the unit located at the margin – that guides choices (and defines price). This leads to the construction of a so-called "cardinal" measure of utility that can be measured and compared between goods, assuming that the consumer is capable of making an evaluation of the utility provided by any combination of goods. However, note that there is no objective scale for measuring utility. At the start of the next century, the *New Welfare Economics* distinguished more clearly between the issues of effectiveness and those related to the distribution of incomes[1] and dealt with them separately. Utility was now considered to be an ordinal measurement which did not allow for direct individual comparisons[2] and questions of effectiveness were evaluated using the yardstick of the Pareto criteria[3] and the Hicks-Kaldor compensation tests[4]. In other words, the question the value is determined less by measurements than by comparisons. The questions of distribution are taken into account by means of the specifications of social well-being functions. Thus, welfarism seems to be more general than utilitarianism, for which social well-being is a simple addition of individual utilities.

---

[1] The distributive aspects led to the construction of social well-being functions that assume the possibility of summing together individual utilities. Without going into the sometimes very technical aspects of this question, we can illustrate it by opposing the social well-being functions of Benthamite utilitarianism, characterised by the sum of individual utilities, and the *maximin* criteria, proposed by John Rawls, which measures social well-being by only focusing on the utility of the least well off (one distribution takes precedence over another if it improves the situation of the least well off).

[2] Ordinal utility only allows for the classification of sets of goods and service by representing them on "indifference curves", constructed in the goods and services space, that plot the set of consumption baskets between which the agents are "indifferent" because they procure the same level of utility. Within the framework of ordinal utility, the consumer is asked to be able to reasonably classify the goods or baskets of goods according to the utility provided. Therefore, he must be able to reply to the question of knowing whether he prefers $q_A$ to $q_B$, $q_B$ to $q_A$ or if he is indifferent to both. In mathematical terms, this amounts to describing a complete preordering covering the set of baskets of goods: the preference relation associated with this preordering must be *complete* (any pair of baskets can be compared), *reflexive* (a basket is preferred to itself) and *transitive* (if basket $A$ is preferred to basket $B$ and basket $B$ to basket $C$, then $A$ is preferred to $C$). Therefore, we construct a mathematical function $U$ from the set of goods to the set of positive real numbers $R^+$ such that: $U(A) > U(B)$ implies that basket $A$ is preferred to basket $B$. We can also construct indifference curves grouping together the baskets between which the consumer is indifferent when he compares them two to two. Due to the *completeness* and *transitivity*, we can then classify these curves according to a total order that is easier to use.

[3] A situation is judged to be optimal, in the Pareto sense, if one agent's utility cannot be improved without damaging the situation of another. This criteria is used to separate – conceptually at least – the question of effectiveness (all of the resources are used to produce the maximum well-being) from that of justice (there are as many Pareto optima as there are initial allocations of resources).

[4] According to the compensation principle, one situation is preferable to another if the agents that benefit from the change can, whilst remaining winners, propose compensation to the losers that leads them to consider the new situation preferable to the old one. By doing this, an indirect form of cardinality is reintroduced into the choices.

We then get the kernel of the "neoclassical" theory of well-being: those goods that have value are those which are useful – that contribute to well-being – and rare, in the economic sense – i.e. those for which the demand exceeds the free supply. This definition of value does not resolve all of the problems. Utility refers back to the preferences of agents, assumed to be pre-existing and stable, but which are only expressed in concrete choices according to the techniques available, the institutions and the social norms (for example, see Elster, 1989). Rarity depends on the conditions for expressing demand and its assessment is therefore dependent on the institutional context[1].

Thus, the economic notion of value reflects not only the more or less necessary or desirable character of objects but also the difficulty or cost of obtaining them. It also enables us to resolve multiple problems, like the classical water-diamond paradox. The marginal value of water may be low whereas its total value is undefined and undoubtedly considerable. Obviously, this question is of central importance for biodiversity.

### *b. The criticisms of welfare economics*

We should mention here that there are alternatives frameworks, sometimes called "post-welfarist" because they start from a criticism of the welfarist premises. The work of J. Rawls (1971), in particular, attempts, from a Kantian point of view, to characterise the basis of a just society as a contract between free, rational and "impartial" people[2]. It shows that unanimity should be reached on two principles: 1. Any person should benefit from the maximum freedom compatible with similar freedom for others; 2. social and economic inequalities should (a) contribute to the benefit of all and (b) be linked to situations or functions open to all. Therefore, this "different principle" stipulates that inequalities are only justified if they improve the position of all (Pareto-improvement) or, at least, if they benefit the least well off. It led economists (Stiglitz, 2000) to construct social well-being functions determined solely by the well-being of the least well off: the social value of a change is then measured by its impact on those that have least (*maximin* principle).

The libertarian criticisms of this approach, formulated by R. Nozick (1974), in particular, consider the situation is just if the procedure that led to it is just (original acquisition, transfer of ownership, compensation for injustices). They put the accent on the freedom of choice principle (present in Rawls) as a more basic value than utility or social well-being.

One of the economists who has discussed the utilitarian basis of economic analysis is A. Sen (particularly 1977, 1987), who proposes 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

---

[1] A pertinent indicator of increasing rarity would be *a priori* an increase in the price, because an economic price is a supply and demand adjustment variable. But the existence of a price assumes that the goods are owned and exchangeable, which is not always the case, particularly for natural assets that have no prime owner, contrary to goods produced by humans.
[2] Impartiality is made possible by the "veil of ignorance" which stops individuals in their "original position" (prior to any contract) from knowing their innate characteristics (sex, race and intelligence) as well as the position that they will occupy in society.

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[1]. This theory of "capabilities" introduces the capability of agents to freely make pertinent choices into the analysis, which is a way of reintroducing the question of inequalities, particularly in development economics.

Therefore, the idea of freedom of choice is present in very different forms depending on the authors and currents of thought to which they belong. In the case of biodiversity, these approaches seem to be attractive because, given our level of ignorance about the state, working and effective services provided by ecosystems and biodiversity, basing our choices on these approaches that are more comprehensive than utilitarianism or welfarism could prevent us from taking potentially irreversible paths.

Another set of criticisms of welfare economics call into question the legitimacy of basing the social value of goods and services solely on the preferences of agents. We should mention the economic psychology work (Tversky and Kahneman, 1981; Kahneman and Tversky, 1982) on this question, which has highlighted the multiple biases (particularly related to the way in which the choices are presented or "*framing*") which may affect the formulation or expression of preferences (see also *infra* the notion of "merit goods"). In the field of the environment, certain analysts' unwillingness to consider that individual preferences constitute a pertinent indicator has given rise to an abundant literature (see, for example, Vatn and Bromley, 1995; Spash and Hanley, 1995; Farber *et al.*, 2002; Sagoff, 2004; 2008) which opposes, in particular, the notion of "intrinsic value", which does not refer to utility, to this instrumental approach to value. This opposition is one of the main points of confrontation between environmental economics, which is comfortable with the pre-eminence of preferences, and ecological economics, which rejects it (Martinez-Alier *et al.*, 1998), and attempts to set up alternative measurements and indicators.

However, the analyses of post-welfarist or ecological economics have not reached the level of development where they can articulate a set of steps involved in making the transition from a theory of the justification of choices to the instruments for analysing a concrete situation. This is why we are going to continue with our discussion in order to make further progress in drawing up a conceptual framework for evaluating biodiversity and ecosystem services within a welfarist framework, without forgetting about alternative approaches or underestimating the limitations of the one we are using.

## 1.2. Rationality and economic effectiveness

Utilitarian and *welfarist* theories offer a coherent framework, from a theory of value to practical tools for evaluating choices or simulating the effects of the political instruments that can be considered, on the basis of a model of the economic agent (the units that may make choices of actions: individuals, households and businesses). The individual behaviour of agents is assumed to be guided by a rationality that

[1] From a similar perspective, Harsanyi had already distinguished preferences for baskets of consumables from preferences concerning the economic institutions that determine upstream the possibilities of existence of those baskets.

enables them to classify the imaginable different states of the world according to an order of preference. Therefore it is a question of an instrumental rationality that cannot lead to a judgement being made on the purposes of the agents: the individual is considered *a priori* to be the best judge of his or her preferences.

### *a. Collective choices and individual preferences*

The notion of preference is central in the rationality-value relationship, but it is a conceptual device because preferences cannot be observed directly. What can be observed are choices and behaviours. Rational choice[1] corresponds to a maximisation of utility calculation[2]. Therefore, a choice is technically is an arbitration between the consumption of one or more goods according to the marginal utility that the consumption of an additional unit of one or the other would provide. It can then be shown that when utility is maximal the ratio of the marginal utilities is equal to the ratio of the prices. The measurement of relative prices does not imply a resort to money because they can be expressed in terms of "opportunity cost": to benefit from an additional unit of one of the goods, the consumer must, *a priori*, renounce a certain quantity[3] of the other (unless it is freely available and free of charge) because the agents only have a limited budget.

It is this constraint on the budgets of agents that makes the link between the markets. Without going into a lecture on economic science, we should remind ourselves that the general equilibrium (simultaneous equilibrium of all of the present and future markets) is the reference model used to judge the efficiency of situations and choices. The two theorems of welfare economics that establish the equivalence between an efficient allocation of resources and a competitive equilibrium[4] imply numerous restrictive hypotheses. Not respecting them generates inefficiencies. Especially, they assume the absence of "market powers" (monopolies, monopsonies, etc.), perfect information on the goods and services and on the preferences of other agents and the absence of externalities, meaning transfers of value that are not the subject of compensation by the market or regulation by another institution. Several of these

[1] In the 1940s, Milton Friedman proposed considering that preferences were revealed by effective choices. This explanation was criticised as being tautological: agents choose what they prefer and the analyst knows that they prefer it because they have chosen it. Even though this explanation is tautological, it nevertheless retains heuristic power, in the same way as the survival of the fittest in the biological theory of evolution.
[2] The link between the microeconomic theory of the consumer and utilitarianism is not direct. It is a descriptive theory according to which the consumer attempts to obtain maximum satisfaction from his or her consumption, personal and not social utility. The link is made through wider theoretical frameworks that include the working of the markets (the "invisible hand"), the rights held over the assets and other institutional and social factors.
[3] Consuming an added unit of the first goods implies an added expenditure equal to their price, with unchanged income, therefore, the expenditure dedicated to the second must be reduce by the same sum, i.e. the agent must forego consuming a quantity equal to the price of the first goods divided by the price of the second.
[4] We can formulate this equivalence roughly as follows: resources are optimally allocated when their allocation results in a competitive equilibrium (decentralisation) and, reciprocally, any efficient allocation can be decentralised in the form of a competitive equilibrium in return for lump sum transfers (which requires the State's intervention in the economy, under several restrictive hypotheses and, in particular, that the forms of intervention do not distort the perceptions on which agents base their choices).

points deserve further development because they can be used to specify the problems raised by the evaluation of biodiversity and ecosystem services.

A central question in philosophy and economics is how to make the transition from individual preferences to collective choices. It is at the root of the social choice theories, which have developed in their modern form from Arrow's Impossibility Theorem[1], which establishes that there is no indisputable social choice function (in other words, we cannot define general interest on the basis of individual preferences). The only general case where the function exists corresponds to a situation in which these preferences can be materialised on a single axis. Therefore, we could define democratic choice procedures to aggregate the citizens' preferences in terms of biodiversity and ecosystem services if we had a one-dimensional measurement of biodiversity or those services and that measurement was commensurate with the other dimensions of the choice (as we can do with a measurement in monetary terms, for example).

We mentioned that the economic theory of value considers *a priori* that the preferences are fixed and exogenous. The basic premise that agents have the ability to classify all of their choice options implies that they know them and that they understand the nature and the consequences. This assumption is never truly realistic[2], but it becomes embarrassing if the problem is managing assets which effectively contribute to the well-being of the agents without them having precise awareness of it. This problem arises in a very evident way for assets that are barely perceptible such as the ozone layer, whose existence most people only learned about when there was controversy about the control of substances that damaged it. But it also arises in a comparable way for the climate and biodiversity, which, even though they can be commonly perceived, are the subject of threats and developments whose perception implies a sophisticated scientific construct. In these situations, agents' preferences revolve around a "mediated" object, i.e. one constructed by others (scientists, experts, journalists, politicians, etc). Therefore, they are contingent upon a set of complex social processes that undoubtedly go far beyond the economists' simple model of instrumental rationality.

### *b. Externalities*

If the optimal allocation of resources corresponds to the equilibrium of a complete set of markets, then the existence of direct interdependencies – not mediated by the economic institutions – between the utility functions of agents will lead to a loss of social efficiency. The notion of external effect (EE) or "externality" is linked to methodological individualism which postulates the interdependence of the choices of the agents. There is an externality when an agent's action influences the well-being of another agent, without that action passing through a market or another regulatory mechanism.

---

[1] The impossibility theorem is a generalisation of Condorcet's Paradox according to which when the choice offers more than two possible issues, a vote does not necessarily result in a majority choice. As a simple procedure no longer offers a guarantee, the justification of social choices implies more complex procedures, involving, for example, the intensity of preferences.
[2] The manipulation of information by interest groups is obviously not specific to current environmental problems. Here, the situation seems particularly sensitive because it is a question of finding information that may be used to legitimate a collective choice in the value of an asset.

Use of this notion for analysing environmental problems was proposed by A. C. Pigou (1920) who suggested their correction by taxes aimed at raising the private cost (PC), supported by the agent who makes the choice to level of the social cost (SC) supported, due to that choice, by all of the agents (formulating the equation: SC = PC + EE). If every decision-making centre had to take into account and pay all of the costs generated by its choices then we would get back to the properties of efficient allocation. In practice, social management of externalities is often organised by law, regulation or incentive mechanisms and therefore by public decisions.

The existence of externalities can be analysed as an imperfection or incompleteness of property rights. This analysis refers to an ideal world – from this point of view – in which every source of social value can be appropriated and is at the base of Coase's conjecture (1960) which proposes resolving the problems posed by externalities by eliminating them, i.e. by defining new and unambiguous rights to the assets concerned and letting agents freely negotiate and exchange those rights. Even though its practical consequences are quite limited once there is a legitimate authority that may implement public policies (the practical relevance of this approach is stronger for international questions), this framework constitutes a reference for the economic analysis of environmental problems[1].

### *c. Public goods*

Pure public goods are characterised by two properties: firstly, it is impossible to exclude use, the goods are available and everyone can benefit from them, and, secondly, the lack of rivalry, consumption by one agent does not deprive the others (national defence is a classic example). In the case of goods with a positive value, everyone wants to benefit from them but no one wants to make an effort to produce them. This is what is called the problem of the "free rider". Therefore, in an economy without public intervention, there is classically under production of public goods in relation to the level that would optimise social well-being given the initial resources. If a public collectivity wishes to remedy this situation, it must, on the one hand, define a procedure for setting the production objective and, on the other, set up the mechanisms leading to the production.

In practice, the real goods that have a public goods dimension are only partially close to this model. The "excludable" goods, without consumption rivalry, are generally called "club goods", because the setting up of a closed group of beneficiaries is an incentive for each member to contribute to their production. The "non-excludable" goods with rivalry of consumption are called impure public goods or common goods; they correctly convey the situation of resources in common ownership when the access rights are badly controlled.

[1] Many factors, which are only superficially resumed by the notion of costs of transaction, constitute limitations for its application to the conservation of endangered species (Moran, 1992). But the question of rights to genetic resources, notably rights of access, is an essential issue of the Convention on biological diversity (Trommetter, 2005). Furthermore, the idea of implementing partial decentralisation of biodiversity conservation policies by creating transferable permits must deal with the fact that the advantages of conservation are not independent of space and that the allocation of habitats leads to heterogeneous costs due to the fact that there are spatial interactions (Drechsler and Wätzold, 2009).

**Table V-1: Exclusion and rivalry of use of economic goods**

| | Exclusion | Non-exclusion |
|---|---|---|
| Rivalry | Private goods | Common goods |
| No rivalry | Club goods | Pure public goods |

*Source: CAS, Groupe biodiversité*

Whether goods belong to one category or another depends on their nature, the social institutions and the technological capabilities. As Fisher *et al.* (2009) for example, point out, ecosystem services can change category depending on the pressure exerted on the ecosystems.

### *d. Common goods*

Many natural resources are resources "in common ownership", which corresponds to the notion of "common goods". This situation gives rise to a real confrontation between reports of inefficiency and crises, often characterised as the "Tragedy of the Commons" (Hardin, 1968), and the work that stresses, on the contrary, their flexibility and adaptability in the face of change (Ostrom *et al.* 1994)[1]. Without caricaturing their position, the opposition partly reflects the difference in perception between the analyses that consider the resources with free access (Cornes and Sandler, 1983) and work concerning assets which, even though they have not been clearly appropriated by people, are nevertheless under the effective control of groups that have defined collective management rules, like the communal or sectional pastures in mountain areas in France, and multiple situations in which resources are under the control of communities on all of the continents.

Goods under common ownership are a legal reality[2]; public goods are a concept of economic analysis that is more or less well reflected by the characteristics of certain goods and services whose production is not ensured in an efficient way by private economic agents. Sandler and Arce (2003) proposed a practical distinction: common goods have the characteristics of public goods but the benefits that can be gained from them can be privatised. Here, we could say, with J. Weber, that biodiversity is a common good and that its conservation is a public good.

More recently, the category of global public goods or *Global Commons* has emerged as an instrument for analysing planetary environmental issues such as climate change and biodiversity (but also humanity's cultural heritage, peace and the stability of agricultural prices). Apart from the fact that this view puts States in the same category as "free riders", incapable of producing goods, which makes a "right to intervene" legitimate again, this category raises particular difficulties due to the absence of a public authority able to effectively ensure their protection, but also due to the fact that

---

[1] We should mention that work on the notion of adaptive management done by the *Resilience Alliance* (www.resalliance.org/). See, for example, Carpenter et Brock (2008).

[2] Here, we should stress that there is a divergence between the legal notion of *res communes* which in civil law designates things which, by their nature, cannot be appropriated, and the economic notion *common goods* which designates goods managed collectively by a group. Part of the Anglophone economic literature also uses the notion of "*commons*" in the sense of the legal notion of *res nullius*, things without an owner but which could be appropriated.

part of the costs and benefits related to them concern future generations. The mechanisms for contributing to the production of these goods remain contingent on the outcome of international negotiations and the goodwill of States to implement their undertakings. Applied to biodiversity, this framework results in the recognition that the rules and actions must be formulated in such a way as to allow for the rationalised preservation of the local and global benefits attached to its conservation (Perrings and Gadgil, 2003).

### *e. The case of merit goods*

In the case of certain goods, the assumption according to which the agents are the best judges of their interests poses a problem. The agents may have reasons for not being the best judges of their own interests, for example, because they are not familiar with goods, because they do not have the means to judge complex issues or because the consumption or use of these goods disrupts their faculties of judgement[1]. The literature designates these goods by the name of "*Merit Goods*" (Musgrave, 1987) because it seems clear that the principle of consumer sovereignty is called into question and that the decisions must be taken on the merits of the goods and not the consumer's ability to pay by an authority which *a priori* a public authority.

The fact that agents' preferences cannot be relied on and that therefore the idea of the preferences of elites, perhaps based on the judgement of experts, is more or less explicitly introduced calls into question the pertinence of the utilitarian approach to social choice. As Brahic *et al.* (2008) pointed out, today, the notion of merit goods has been legitimated by certain developments in the standard theory, such as behavioural economics, and even more for all alternative approaches that take into account the social determination of people and their behaviour.

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 rider"). The two categories often overlap (Head, 1974; Fiorito and Kollintzas, 2004) and this is the case for biodiversity. 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. The agents' relationship with the public authority can therefore be considered to be a delegation of choice[2].

Faced with this difficulty – and for as long as the situation is not the subject of wide social consensus (like obligatory schooling, vaccinations and the banning of hard drugs) –, the idea of constructing a measurement of the goods to be produced or preserved and to choose it commensurate with the other consumer goods could however form a useful step towards taking these goods and a practical means of socially managing their production into account[3]. This finding invites us to look closer

---

[1] J. Elster (1979) illustrated these limitations of rationality by the metaphor of *Ulysses and the Sirens*, which led him to define a " second order rationality".
[2] To justify this delegation of choices to an authority that is recognised to have better abilities, we can mention that the notion of "commitment" defined by Sen (1977) to qualify the state in which an individual resists his or her preferences to realise his or her meta-preferences.
[3] In the same sense, the values determined by the State's services to take such issues, which market mechanisms handle very badly, into account in public economic calculations are called "merit values" (see CAS, 2008b).

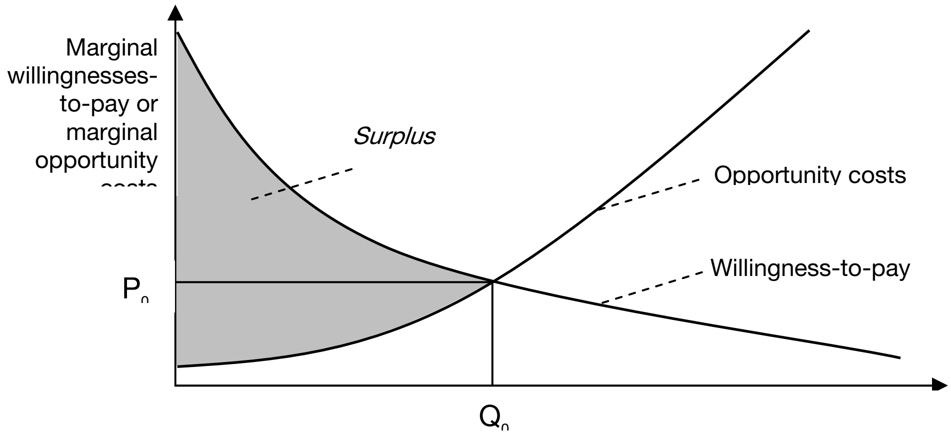
at the challenges of a monetary measurement of values that escape from market exchanges and, as we will see, do not necessarily result in use behaviours.

## 1.3. Value, price, money

### *a. Value and price*

The value of assets is linked to their marginal utility. More specifically, their social value is the sum of the marginal utilities of each unit for all of its users. This marginal utility can be approached by the agents' maximum willingness-to-pay to obtain additional units of the goods. If we classify these willingnesses-to-pay in decreasing order, we can trace a curve such that the area under that curve is the total value of the goods. If those goods have an opportunity cost (related to their production or the fact that consuming them today will mean that no benefit can be derived from them tomorrow), then the area between the costs curve and that of the willingnesses-to-pay represents the surplus (the surplus value from which the agents benefit after having supported the costs).

**Figure V-1: The surplus as an approach to the social value of goods**




*Source: CAS, J.-M. Salles 2009*

Point $Q_0$ defines the optimal quantity of the goods that should be produced because beyond this point the production of additional units has a higher cost than the agents' willingness-to-pay. If the goods are the subject of transactions, the ordinate $P_0$ of the point of intersection of the curves at that abscissa is the price at which the exchanges will theoretically be made. The horizontal right segment between this point and the axis of ordinates separates the part of the surplus that will be appropriated by the producers (below) from that which will fall to the consumers.

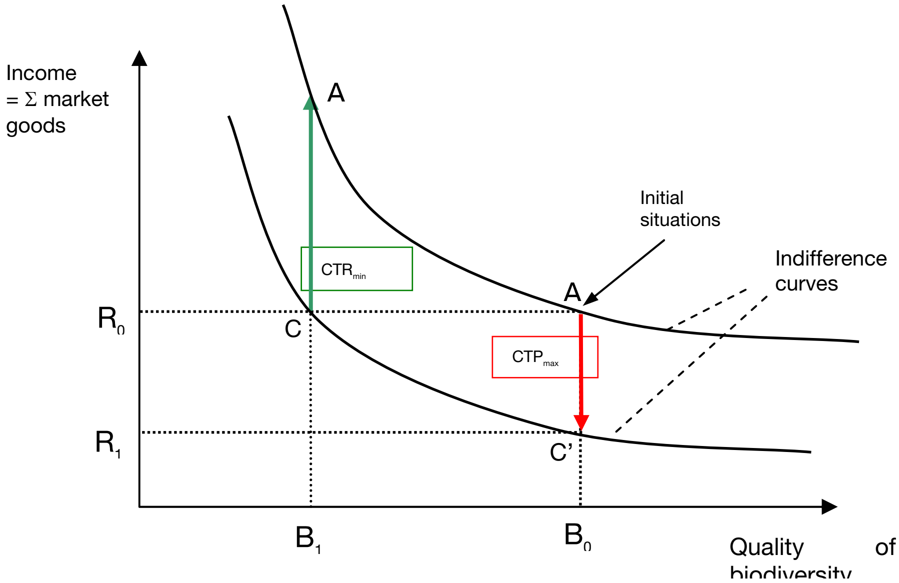
The theoretical framework underlying the evaluation of natural assets is clearly exposed in numerous manuals[1] and we will only repeat the structural elements here. The exact measurements of surplus have been refined [2] and, in order to be consistent with the other economic choices, the evaluation must be based on the Hicksian notion of demand which results in a compensated measure of surplus, so as to keep the level of utility constant. The variation estimated from the initial level of utility (before the change) is called the *compensatory variation* of surplus and that estimated from the final level the *equivalent variation*. However, we should point out, after Willig (1976) that, for most practical choices, the measurements of these different concepts result in differences of less than 5% (less than the margin of error).

---

[1] See, for example, Perman *et al.* (2003) or Bonnieux and Desaigues (1998), for a clear and rigorous presentation in French.

[2] The simple presentation that we have just made assumes implicitly that the marginal utility of one Euro of additional income is constant. In other words, the improvement in consumer satisfaction is the same if his or her income increases from 10 to 11 Euros as if it increased from 1,000 to 1,001 Euros.

**Figure V-2: Willingness-to-pay or receive for a degradation of biodiversity**




*Source: CAS, J.-M. Salles 2009*

Figure V-2 gives a very simplified presentation of the main situations that may result from a variation in the quality of biodiversity starting from a case of degradation. The agent's initial situation is at A, benefitting from biodiversity $B_0$ and having revenue $R_0$. A project threatens to degrade biodiversity to $B_1$ and the agent has the possibility of intervening in this choice by offering to pay a sum aimed at financing a modification of the project so that it no longer has an impact on biodiversity (from the agent's point of view). The effect of the project would be to change the agent's situation to C. It would therefore be rational for him or her to pay any sum less than that which will take him or her onto indifference curve $U_1$. Therefore, payment AC' that results in a reduced income $R_1$ (equivalent variation) is his or her maximum willingness-to-pay. This situation assumes that the agent had no right to the biodiversity, his or her sole means of influencing the decision being to incite the project's promoter to modify it. If, on the contrary, he or she has legally defendable rights, the promoter must offer compensation (compensatory variation) at least equal to the increase in income CA' that enables him or her to remain on indifference curve $U_0$, known in French "consent to receive" and English as "*willingness to accept a compensation*".

In real measurements, the divergences between WTP and CTR is sometimes very large (up to a factor of 10) and is sometimes interpreted as a sign of the irrationality of agents or the lack of robustness of the methods. However, there are good reasons for this divergence, related to the income effect (WTP is dependent on budgetary constraints whereas the CTR is not and agents tend to overvalue losses in relation to gains) and the imperfect character of the substitution of market goods for natural assets (see, in particular, Hanemann, 1991; Shogren *et al.*, 1994), in particular, in the face of irreversible losses. Many authors (especially Arrow *et al.,* 1993) conclude that the WTP measurement should be prioritised because it better reflects a budgetary arbitration choice.

We must stress one essential point: tracing an indifference curve in the goods space (here, market goods and the quality of the biodiversity) implies that the goods are substitutable in the agent's utility function, i.e. that by proposing choice situations to the agent we can determine an increase in the quantity of one of those goods which results in a gain in utility that compensates for the loss of utility linked to be the loss of a unit of the other goods. This is obviously a major hypothesis that may go against common sense if one of those goods is the support of ethical or intrinsic values. There is a response to this objection, called "lexicographical preferences", which consists of considering a hierarchy in preferences (like the order of letters in a word can be used to classify a lexicon)[1]. If, for an agent, preferences concerning health, for example, dominate those concerning cultural consumption, that agent will be ready to renounce any opportunity in the cultural field to improve, even very marginally, his or her health. This question deserves to be asked in the field of biodiversity. Nevertheless, it is probably true that only catastrophic consequences in the evolution of biodiversity or the demonstration of threats to the services vital for humanity would justify conserving it "at any price".

There is a direct link between "marginalism" and evaluation. This is why the analysis framework is only *a priori* robust for small variations. As we have stressed, only the behaviours are observable: here, the point of intersection of the curves that defines both the quantity exchanged and the price of the transactions. The remainders of the curves are not known and the purpose of the evaluation methods (see below) is to make them in part observable. These curves are, above all, hypotheses that explain behaviours… if there are observable behaviours (there are no direct ones for goods that are never the subject of transactions). If the changes have structural effects, the reference data may no longer have any sense: we can, no doubt, evaluate, at least in part, the services provided by a hectare of forest, but extending the result obtained to all of the temperate forests would raise considerable problems.

### *b. Value and money*

Economic evaluation does not necessarily imply monetarisation. Up to now, the magnitudes in question (utility, well-being and opportunity cost) have been measured relative terms. Economics compares but is bad at measuring. To transform that into a measurement, we must define a unit and the reference to willingnesses-to-pay, to "prices" that are observable indicators, makes us think that the unit of measurement will be money, in its function as a general equivalent basis. This choice can be justified by the simple fact that agents and public decision-makers spontaneously perceive the significance of monetary measures, relative to their income or the price of certain market goods.

The neoclassical concept of value (see above) is, *in fine*, defined as the monetary compensation that maintains indifference. Money appears here as a simple unit of measurement with the dimension of price; which does not imply the next step, i.e. the obtaining of economic prices on a market. However, in practice, the use of money as a unit has several important disadvantages.

1. Money is not a unit that is stable over time. The value of money varies with economic policies and all of the factors that influence inflation. Therefore, the use of a monetary unit implies the specification of a reference year (1 euro$_{2004}$ •

[1] Taking lexicographical preferences into account implies *ad hoc* and difficult treatment in the evaluation methods (see Rekola, 2003).

1 $\text{euro}_{2008}$), unless all of the elements are simultaneous or unless, if the subject does not involve comparisons over time, prices at current value can be used.

2. Money is not a unit that is stable in space. Many of the results in the literature are denominated in US$ whose exchange rate in Euros or in any other currency fluctuates for multiple reasons, making international comparisons difficult, even if corrected by "purchasing power parities" which attempt to adjust the currencies to their consumption basket equivalent.

3. Money is not a unit that is stable in society. When it is a question of measuring variations in surplus linked to a change in the availability of an asset, the marginal utility of the money is not the same for everybody (it is *a priori* greater for those with the lowest incomes). The gains or losses in social utility depend on the distribution of income and the monetary yardstick tends to worsen the under evaluation of assets that benefit the poorest people.

These points can combine together and we can easy understand, to take a controversial example, that a poor African subsistence farmer's willingness-to-pay to reduce the number of elephants threatening his crops cannot be compared, without precautions, with that of rich foreign tourists who want to see them multiply in the reserves. Conversely, the fact that there is a difference of several orders of magnitude between, for example, the value of protecting the mangroves in Malaysia and in Florida is not mainly explained by a monetary effect but rather by the gap that exists between the income levels in those two states and the presence of parts of the heritage, especially buildings, that are actually protected by these coastal ecosystems.

Along with the technical difficulties that we have mentioned, money can also be the subject of hoarding or speculation and thus see its role of general equivalent undermined. Therefore, its use as a yardstick for value can raise many difficulties or problems and the decision-makers may employ alternative solutions to the monetarisation of the benefits, a summary description of which we will give below. We should first specify the relationships between evaluation and public decisions.

## 1.4. Evaluate for better decision-making?

Economic evaluations are generally motivated by the prospect of decision-making: choosing a project, for the *ex-ante* evaluations, or determining the indemnities using *ex-post* analyses. The fact that the relationship between evaluation and decision-making is mediated by the reality of public decision-making procedures should incite us to better understand that relationship (in chapter VII) and, in the immediate, to specify the status of reference values used in the socio-economic evaluation of projects.

### *a. One prior question: from what perspective are we evaluating?*

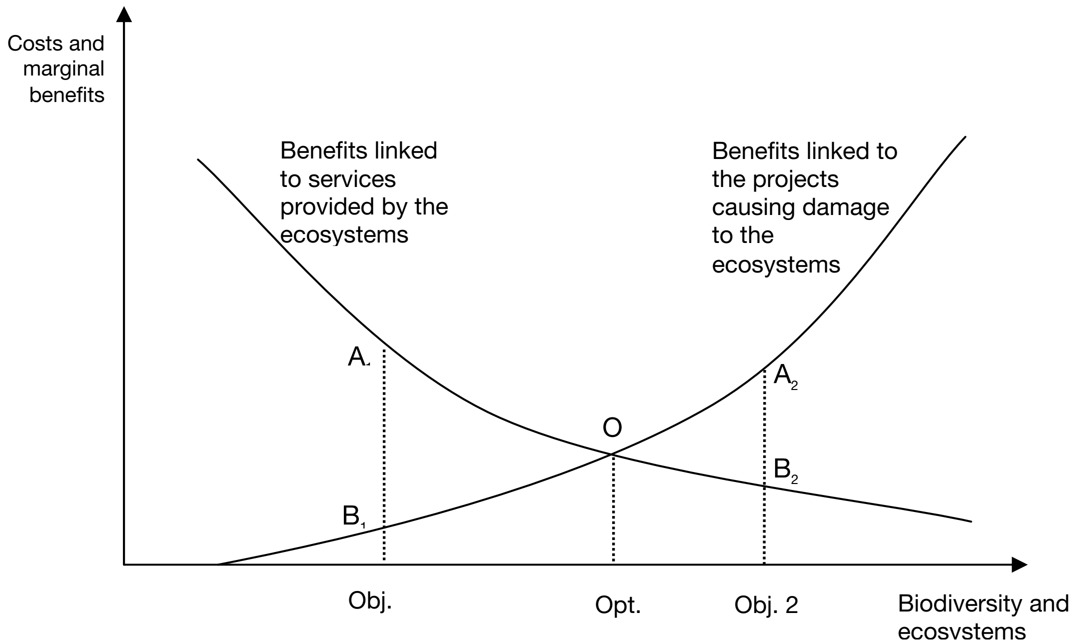
If we are evaluating biodiversity and ecosystem services, it implies that we must place ourselves, in a simplified way at least, within scenarios that include hypotheses about the development of interactions between human activities and ecosystems. These scenarios depend on knowledge about the dynamics of more or less human influenced ecosystems and hypotheses about the development of human activities. They may include stated objectives in terms of conservation of biodiversity.

When evaluating the issues involved in a public decision or a project subject to authorisation, two perspectives are likely *a priori* to be retained:

- The case of a policy that aims to attain a given conservation of biodiversity objective at the least cost; in which case, only the costs corresponding to the achieving of that objective have to be estimated, using a cost/effectiveness analysis whose purpose is to compare the marginal gains obtained for the expenditure of one euro on the different actions that may contribute to the objective set.
- The case of a so-called "optimal" policy characterised by the fact that the cost of the marginal effort of conservation is equal to the marginal value of the biodiversity preserved. This is a scenario in which we place ourselves, at least implicitly, in a cost/benefits analysis. The objective here is to ensure that the costs supported to conserve biodiversity provide a marginal gain of well-being equivalent to the expenditure committed in other fields.

We can show these two options in a simplified diagram: point Opt. represents the optimal policy; points Obj. 1 and Obj. 2. are two other possible objectives of policy. If the objective is not very ambitious in terms of conservation (Obj. 1), then we renounce all of the services located to the right of this point (therefore the benefits provided by each additional unit of biodiversity or ecosystem that is less than $A_1$), but we can carry out all of the development projects providing a benefit, even quite slight (greater than $B_1$), located on the left of this point. Reciprocally, if we want to attain a high conservation objective (Obj. 2) then we benefit from all of the services with a value greater than $B_2$ and we have to renounce all of the projects that do not involve the destruction of a unit additional to at least $A_2$.

**Figure V-3: Value of ecosystems and costs of conservation**




*Source: CAS, J.-M. Salles 2009*

In the case of an optimal policy, the surplus of benefits linked to the ecosystems relative to the costs of conservation is maximal (by definition). If the objective set by the policy differs from the optimum, then the marginal costs supported will be different from the marginal benefits obtained. Objective 1 entails renouncing surplus $A_1B_1O$ whereas objective 2 implies supporting costs $A_2B_2O$.

### *b. Optimal conservation policy*

From an economic point of view, conservation policies should be analysed as an instrumental objective amongst the set of means that contributes to improving the social well-being. Only this analysis would enable us to define an "optimal conservation policy" that corresponds to the situation where the least cost policy objective corresponds to the social optimum of conservation. A possible representation is a budget allocation problem (Wu and Bogess, 1999) which is not forcibly monetary and must include the opportunity costs, among several objectives. To be fully rigorous, this analysis should be done within the framework of general equilibrium, because we must consider that the choices made in this field will have an impact on future wealth and therefore on the size of the budget which may be allocated to conservation in the future.

Figure V-3 is a simplified representation of this objective corresponding to the situation in which the two values Ai and Bi meet at the optimum O. Though this is still an inevitable reference point, the practical determination of the optimum is not a realistic objective and it is even difficult to know on which side of the optimum we find ourselves. Therefore, the question of the objective in relation to which biodiversity and ecosystem services should be evaluated needs to be answered.

### *c. Least cost conservation policy*

This similarity between their names sometimes makes people think that cost/effectiveness analysis (CEA) is a simplified form of cost/benefits analysis (CBA), but they are of a completely different nature because CEA aims to determine how to attain an objective that is not subject to evaluation effectively, i.e. by minimising the losses of social well-being. We can associate CEA with approaches of the *As Low As Reasonably Achievable* (ALARA) type that do not generally include explicit calculation of the economic and social benefits expected from the objectives pursued. This CBA-CEA difference is of real practical importance for setting the reference values for biodiversity and ecosystem services. In practice, from a CBA point of view, having values for biodiversity enables us to calculate to what point protection operations are economically justified. From a CEA point of view, the reference values are used to compare the relative effectiveness of a set of protection operations or measures. The analysis does not aim to define the size of the optimal effort but its optimal distribution in a basket of possible measures.

Accepting the idea that the budget allocated to conservation is limited, a philosopher like Norton (1987) thought that a cost/effectiveness criterion could be applied, considering that the definition of the objective could escape from value judgements by being based on a criterion that he qualifies as "formal"[1] such as richness in species or maintenance of a diversified gene pool.

This approach could take on increasing importance if a "compensation market" was to develop. As Géniaux (2002) pointed out, the notion of compensation contains the idea that conserving or restoring nature does not entail the same costs everywhere, particularly in terms of opportunity costs. Even if compensation is restricted to an ultimate solution, after the possibilities of preventing the deterioration and minimising the damage have been implemented, its logic is that, once these steps have been taken, there may be residual damage less costly to compensate for than to prevent. The function of a "compensation market" would be to ensure the identification and effective compensation, by the restoration of damaged ecosystems not just the protection of existing ecosystems, in situations where the excessive cost the prevention lead to the destruction.

In order for a conservation objective to be attained at the least cost to society, we must carry out as a priority the operations that entail the lowest total conservation costs (including the opportunity cost related to the activities of producing useful goods and services which we must forego). This result could be obtained by including costs for biodiversity and services lost in the socio-economic evaluations leading to the renunciation of those projects (by cancelling their profitability as it is calculated by the net present value, NPV). In figure V-3, to attain the least ambitious conservation objective Obj. 1, that would correspond to taking the value of the ordinate of point $B_1$, therefore to accepting most of the development projects and only foregoing those that only derived slight benefits from the degradation of ecosystems.

Note that the value that should be retained by an authority that wishes to direct all of its development choices towards a given level of conservation is not the marginal social value of the benefits obtained, $A_i$, but the marginal social cost implied by their conservation, $B_i$, i.e. the social valuation of the destruction. Therefore, applied to a country's conservation strategy, this approach would involve having a coherent representation of all of the forces that exert pressures on biodiversity and drawing up scenarios including a description of the state of ecosystems and their likely evolution in the face of those pressures[2].

### *d. Reference values for better decision-making*

When we look at the actual evaluation practices, we are tempted to quote G. Heal (2000): *"If our concern is to conserve these services (ecosystem services), evaluation is largely irrelevant. I would like to insist on a point: in the matter of protecting nature, evaluation is neither necessary nor sufficient. We conserve many things that we do not evaluate and few of those that we do".* In spite of this, G. Heal chaired the Committee on Assessing and Valuing the Services of Aquatic and related Terrestrial Ecosystems, set up by the National Research Council, whose report (Heal *et al.*, 2005) is

[1] Norton opposes "formal" criteria, not involving value judgments, to "basic" criteria that promote objectives like well-being or the survival of Humanity.

[2] For instance, the DPSIR approach whose advantages are well established but the implementation of which would have surpassed the objectives of this report (see chapter VI). See, for example, the feasibility study of an integrated *drivers-pressures-impacts* socio-economic model of biodiversity based on the European research platform on long-term socio-ecological monitoring (Haberl *et al.*, 2009).

entitled *Valuing ecosystem services. Toward better environmental decision-making*. This title could be that of our report. The objective of evaluating the services provided by ecosystems is not necessarily to arrive at the "*best*" decisions but at "*better*" decisions.

We cannot, no doubt, define "shadow values" for biodiversity and ecosystem services in the sense that expression took in the report on *The shadow value of carbon* (CAS, 2008b); i.e. the value selected by the state to harmonise the implicit costs of the effort made to limit greenhouse gas emissions in all of its operations and to achieve a given reduction objective, the optical nature of which will not be discussed here but which is related to France's international commitments.

What we can do, however, is to determine values which reflect the benefits that society enjoys, at the present time and according to a foreseeable scenario, from the uses or the existence of a certain state of biodiversity and the ecosystems. The value of the benefits is the same thing as the costs of degradation. In the context of the cost/benefit analysis of a project, these values reflects the losses related to the project's impacts on the ecosystems and must be deducted from the project's net value or that of its variants being studied. Notwithstanding its practical limitations (Pearce, 1976; Odum, 1982), cost/benefit analysis is still the reference framework (Fisher *et al.* 2009) that economic analysis can use to help us take into account the questions involving biodiversity and ecosystem services in decision-making. But there are alternatives.

## 1.5. Alternative approaches to economic evaluation and monetarisation

Practical considerations (it is not technically possible to obtain usable values) or considerations of legitimacy (values are not socially accepted) may lead us to renounce evaluating the benefits of conservation in monetary terms. Several alternative procedures are possible, each with its advantages and limitations.

### *a. Minimum standards of safety, viability and precaution*

When natural areas are transformed, ecosystems destroyed and biodiversity lost, the consequences are not well known and difficult to perceive whereas they may in the long term threaten fundamental aspects of our existence. This prospect should lead us to adopt a prudent attitude, which may take several forms.

In 1952, Ciriacy-Wantrup suggested **safe minimum standards** (SMS) as a decision-making rule for the management of renewable resources. They have been widely debated, particularly in the United States for the implementation of the *Endangered Species Act* (ESA). The idea is that the conservation of the species is assumed to be beneficial as long as it does not lead to intolerable costs. Some authors (OCDE, 2002) consider that the SMS imply that the burden of proof is on those who wish to increase the risk, for example, by reducing biodiversity. Therefore, it is not a question of demonstrating the benefits of conservation, only demonstrable evidence of the "excessive costs" could lead to abandoning it. In spite of repeated presence in the economic literature (Bishop, 1978; Randall, 1991; Ready and Bishop, 1991; Randall and Farmer, 1995; etc.), the SMS present an analytical challenge, because they introduce an asymmetry between the costs whose status is still to be specified: the assertion that the losses of biodiversity constitute a greater increase in risk is still disputable, even though it is consistent with the idea that the consequences of the destruction, even locally, of things that determine the resilience of ecosystems can have unforeseeable cascading effects (Kinzig *et al.*, 2006).

A group of works on viability and precaution have attempted to explain the reasons behind the advantages of conservation. The approaches in terms of **viability** involve modelling the evolution of status variables (the level of biodiversity or ecosystem services, for example) as a function of its dynamic's parameters and control variables (the pressures exerted by certain human activities on the ecosystems, for example) which may vary within certain limits. This procedure is used to define a domain of viability for the status variables from outside of which the control variables can no longer return them. Béné and Doyen (2008) have shown, on the basis of numerical simulations, that biodiversity (measured by a Shannon index) has a positive effect on the performances of ecosystems measured in ecological and economic terms.

The interpretations of **precaution** are still the subject of controversy, even if we get away from the naive and unusable definition of abstention in the face of uncertainty. Many authors define it by the idea of prudent and reasoned choices when faced with uncertainty. Such a definition does not give a precise rule for decision-making but leads to the assertion that it is necessary to organise a regular and interactive process between knowledge and action that should result in a dynamic approach to the measures to be taken[1]. This is the perspective within which economic analyses, like that of Gollier *et al.* (2000), which have clarified the notion of precaution relatively to the anticipation of an improvement in the information, have developed. Henry and Henry (2002) formalised scientific knowledge about the plausibility of *events,* as well as the scientifically unambiguous concepts of events and acts, which enabled them to demonstrate the less than optimal nature of choices based solely on unambiguous information. Henry (2006) clarified the notion of reliable uncertain information, which led him, whilst recognising the difficulty of using decision-making models with an "aversion to ambiguity"[2], to propose choice criteria which, by generalising the von Neumann-Morgenstern criteria, may provide operational decision-making instruments in uncertainty.

Several authors (Pearce, 1976; Page, 1977; Randall, 1991; OCDE, 2002) have suggested that an instrumental approach, like CBA, could be combined with a prudent approach aimed at conserving "the living fabric": the conservation of a minimum level of biodiversity would be a durability imperative but beyond that limit, CBA or other forms of evaluation of equivalence could be applied. At minimum, we can accept that precautionary principle implies that vigilance structures should be set up and evaluation and measurements revised periodically.

[1] "When it is realised that damage, even though uncertain in the state of scientific knowledge, could seriously and irreversibly affect the environment, the public authorities will, within their areas of competence and in order to apply the principle of precaution, make sure that the procedures for assessing the risks are implemented and the provisional and proportionate measures are taken in order to counter the damage done" (article 5 of the Environment Charter, 2005).

[2] Aversion to ambiguity results in differentiated weighting of certain and disputed information (for a simple presentation see Henry, 2006).

### *b. Multicriteria analyses*

Multicriteria analyses enable us to avoid reducing the dimensions of a choice to a single measurement when comparing the different choice options. We define a series of criteria that will be evaluated independently (for example, by using a battery of indicators), then we construct a form of aggregation that can be used to rank the options. The difference from a single measurement lies mainly in two points:

- different criteria may be treated differently, by including qualitative aspects or by distinguishing substitutable criteria (a deterioration of one may be compensated by an improvement of the other) from essential criteria (their measurements must always reach a minimum value),
- the composition function is explicit; the weight of each criteria remains visible, for example, during a process of consultation or negotiation (therefore, it can be manipulated retrospectively according to the choices that are specified as a result of it, which can be seen as an advantage or a weakness).

There are two main categories of methods for composing the weights of the different criteria. The first consists of aggregating then comparing. This is a question of resuming the value of any alternative by a global score v(a) calculated from its vector of performances. This score is taken to summarise the global value of the alternative and is used as the basis for the multicriteria comparison of the alternatives. This way of proceeding is very widespread, for example, in education where the comparison between two students is based on the average of their marks.

The second consists of comparing then aggregating. This is a question of first comparing the performances of the alternatives criterion by criterion. A binary index of partial preference is defined for each alternative and each criterion that is an increasing function of the first argument and decreasing of the second. The preference between the two alternatives is then defined by aggregating the partial preference indexes. In this approach, the same function is used to compare the performances of two alternatives on the same criterion. Two performances associated with different criteria are never compared directly. Therefore, it is not necessary to assume that you know how to compare an alternative's performances on different criteria. Conversely, it is assumed that you know how to compare binary partial preference indexes for two different criteria. This second method is more complex to implement but is considered to be less subjective and less subject to manipulation.

### *c. Analyses based on objective measurements*

A set of methods have tried to dispense with from the subjective dimension of money by defining a more objective units of measurement. They thus leave the field of utility and human values in order to measure physical or biological magnitudes. The three approaches have aroused a certain amount of interest in the last few decades because they responded to expectations. The first is essentially descriptive whereas the two others have more prescriptive ambitions and can at least be used for classifications.

The **material balance** consists of measuring an entity's incoming and outgoing flows (Ayres and Kneese, 1969). Starting from the principle of conservation of matter, this approach can be used both to monitor the flows, for example in a production unit or business, but also identify all of the substances leaving the process and, by the difference, measure *a priori* what will be left in a natural environment. It is an interesting basis for "industrial ecology" type procedures which aim to limit the impact of installations. Even though it provides potentially interesting information, it cannot be used on its own to guide choices because it is not really based on a principle of effectiveness. The minimising of a certain type of effluent is not *a priori* a pertinent objective, unless there is a good reason to think that this form is less beneficial than another. It is partly to answer this question that procedures have been developed which aim to quantify the flows using a common unit, energy.

Based on the ecology of the Odum brothers (Pillet and Odum, 1987), **eco-energy analysis** starts from the fact that energy is a magnitude common to the analysis of ecosystems and socio-technical systems *("energy is Nature's money").* By measuring the flows in terms of energy, we can follow them from one sphere to another and analyse what becomes of them in each. The principles of thermodynamics will provide the analysis framework (or metaphors) aimed at evaluating the relative performances of the different choice options. Several approaches have been studied depending on whether they start from the energy content of the flows ("*emergy*" for *"embodied energy"*) or the free energy that they transport and release ("*exergy*"). This procedure may not be very well suited to measuring the qualitative impacts on biodiversity but it can provide pertinent information for evaluating the quantitative impact of infrastructure or industrial projects on the natural environment, particularly from a comparative point of view[1]. Here, the question is to define a value scale that will include this type of information along with the other dimensions of economic effectiveness and social justice. But the objective information resulting from eco-energy and material balance analyses can take its place among the indicators of a multicriteria analysis.

More recently, W. Rees and M. Wackernagel (Rees, 1992; Wackernagel et Rees, 1996) proposed the notion of **ecological footprint**. This measures the difference types of impacts of human activities, both in terms of the use of space and consumption of resources and the employment of ecosystems functions to treat our waste and pollution, as a necessary surface area equivalent, measured in "global hectares". This approach has been applied to countries, towns, businesses and even an individual. Therefore, it could provide part of a non-money based answer to the question of the impact of projects on biodiversity, if we are able to create a weighting of the surface areas affected by a project according to the services and the biodiversity that they support (Senbel *et al.*, 2003). Abundant criticism has shown up certain weaknesses of measurements of ecological footprint by demonstrating that its use is marred by limitations (in particular, concerning the certain calculation conventions) that make it more a tool for political communication than a scientific concept (Van Kooten and Bulte, 2000; Fiala, 2008). However, the uses made of a tool by "militant" organisations should not discredit its use in the pursuit of knowledge (Foley *et al.*, 2007; Haberl *et al.*, 2007) and for assessing the pressures exerted on ecosystems by human activities. Nevertheless, we can note that, even though the discussions in progress will no doubt

[1] The work which, following the approach initiated by N. Georgescu-Roegen in the 1970s, posed the question of the limitations on the technical effectiveness of economic development, in terms of entropy, can be seen as similar to these analyses, even though they sometimes find themselves in competition, opposing the more radical vision of a "bio-economy", which advocates negative growth and the redirection of development, to ecological economics.

lead to an improvement in the quality of information provided by ecological footprint measurements that does not confer on it *a priori* any normative value[1].

Finally, we should mention the renewed interest in the deliberative procedures in which the perception of problems is constructed with the people concerned or a representative sample of them. These procedures may remain qualitative and aim at mutually enriching the perception of the issues by the people responsible and individuals concerned as in citizens' juries. They may aim to create a cross-fertilisation with the economic evaluation techniques by structuring the exchanges so that they result in the expression of reasoned preferences that may be expressed in monetary terms[2].

## 1.6. To conclude

The objective of these long discussions has been to characterise economic valuation and specify its epistemological foundations: anthropocentrism, consequentialism, utilitarianism, subjectivism and marginalism. The challenge was to make clear the meaning of an economic evaluation of biodiversity and ecosystem services, to explain the advantages and limitations of the use of a monetary yardstick and, finally, to briefly describe what we can expect from alternative approaches.

Within the framework of its premises, economic evaluation seems like a coherent attempt at a procedure for analysing alternative choices. The resulting notion of value mirrors that logic (utility - rarity) and leads to the allocation of an instrumental value to assets that reflects their contribution to social well-being relative to the alternative possibilities. However, its requirements, particularly in terms of information and calculation capacity, may limit the validity of the results obtained.

The central argument in favour of a monetary measurement, commensurate with prices, no doubt resides in the fact that it is a synthetic indicator which includes considerations related to both the utility and relative rarity of the assets. Therefore, in spite of the questions about the effects specific to money, it is directly correlated with the properties on which their social value is based. The alternative approaches also suffer from important limitations from the point of view of their universalism or their consistency with social functioning.

Both the prospect of generalised appropriation of all present and future assets which would leave the monopoly of regulation to the market and the temptation to avoid any explicit evaluation and leave everything to political choices or collective action processes do not seem to be options with general practical applications. However, it is evidence that progress has been made and can be expected in both of these areas:

---

[1] The assertion that the global ecological footprint should not exceed the planet's biological capacity does not stand up to a dynamic analysis. We cannot see *a priori* what the optimal footprint would be for the impact of a local project. The idea of reducing this impact seems *a priori* interesting, but without the minimising of the footprint being necessarily socially desirable, even in the long term.

[2] This has led many authors, like Sagoff (1998) or Tacchoni (2000) and others (see OECD, 2002) to propose "democratic" collective deliberation procedures enabling groups of citizens to explore the questions posed before expressing judgements. See also chapter "Getting values: deliberative and participative procedures" in the OECD manual (2002, p. 79 and following).

- definition of clearer rights to the assets that support biodiversity can decisively contribute to the better taking into account and taking responsibility for its value, without explaining on measuring it, but by clarifying the responsibilities,
- the building of consensus about the importance of conserving some assets is an essential phase, where it is possible, for getting the social players to adopt the collective choices and, even more, getting them to respect and implement them.

But for many local choices whose consequences potentially or really, and by indirect, complex, uncertain and controversial mechanisms, affect heterogeneous players or populations, the limitations of the approaches mentioned become evident. The impact of infrastructure projects on biodiversity is typical of the type of situation in which a clear separation between the explanation of the issues by experts (scientific, technical and socio-economic) and the statement of priorities (social, strategic and political) by decision-makers is an essential part of democracy.

A fundamental characteristic of standard economic evaluation if that is based on the preferences of agents. This is generally its strength: forming a link between the principles of democracy (each voice counts) and the critical base (the weight of each voice is linked to income). But this basis seems weak if the agents have no familiarity with the assets that are being evaluated, as is the case of biodiversity, because the preferences are then based on frail and potentially biased information.

Alternative approaches may have great pertinence in the analysis of the options of a choice. Maintaining ecosystems in good working order is no doubt a question of survival for humanity and there are ethical or moral values attached to nature that it may well be better to deal with by other means than economic evaluation, if we can ensure that this differentiated treatment does not mean that they are marginalised or ignored. The precautionary principle may also lead to the modification of the weight given to certain consequences of choices (OECD, 2002).

Most resources can be the subject of alternative uses. Therefore, their opportunity cost is not null and all of the choices that we make can be evaluated in relation to what they imply that we forego. The search for the best allocation of resources, i.e. the determination of the choices that enable us to create a greatest social well-being, entails a form of analysis of their pertinence for which economic evaluation offers a coherent framework. Even though it raises a set of problems, recourse to the monetary yardstick facilitates comparison with the set of other choices, particularly in the allocation of public finances.

Nevertheless, the explanation of the foundations and significance of economic evaluation has highlighted the importance of attaining dual objectives:

- to construct equivalence classes, because all projects will have impacts on biodiversity and it is important that we are able to compare them and, in the case of those that cannot be avoided in spite of their importance, to compensate for them,
- to define a unit of measurement with a sufficiently wide scope so that, in the same way as money, it can be used as a general equivalent when the

comparisons or compensations cannot be done respecting a multidimensional similarity.

We can now broach the conceptual aspects of economic evaluation of ecosystem services and biodiversity knowing that if their implementation runs into difficulties, certain of its principles will probably also guide the alternative solutions.

## 2. Evaluating biodiversity and ecosystem services involves a widening of the point of view

Immediately the notion of biodiversity emerged, the question of its value was posed. The creation of this word quite evidently implied the project of highlighting the importance of living diversity for human societies. Thus, amongst the 57 contributions to the work published by the American *National Academy of Sciences* (Wilson, 1988), three chapters were due to economists (Hanemann, 1988; Norgaard, 1988; Randall, 1988). Following close on its heels, a complete work was published on the question of incentives for the conservation of biodiversity (McNeely, 1988) which seemed more urgent and, no doubt, more practical. But economic analysis of conservation policies necessarily encounters the question of the value of biodiversity and the services provided by ecosystems.

However, even though the nature and significance of economic evaluation explains the interest of applying it to the choices affecting biodiversity and ecosystem services, they are not sufficient for asserting that it can be used indiscriminately and without specific adaptations. Implementing implies that answers must be provided to three types of questions. Why does biodiversity have a value for our societies? What are the specific developments that enable us to validly apply the framework of economic evaluation to these particular assets? Finally, does the temporal dimension of the relationship with nature have specific implications, in particular for the question of discounting?

### 2.1. Why does biodiversity have a value?

Economic values are instrumental values. They portray the contribution of goods and services to well-being. Classically, we distinguish the private value of goods for an agent from their social value, i.e. for all of the agents in a society. Between individuals and society, there are groups (farmers, hunters, tourists, etc.) whose activities may be directly dependent on certain ecosystem services. Over above an individual society (French society at the beginning of the 21st-century, for example) certain parts of biodiversity could be considered to be a common heritage of humanity and, *in fine*, we must posit the question of anthropocentrism. From the point of view of economic analysis, the question of value implies specifying if the benefits related to the ecosystems are essential, irreplaceable or substitutable, which will lead us to asking ourselves about the economic nature of biodiversity and ecosystem services.

#### *a. The question of anthropocentrism*

Can the value of nature be analysed from the sole point of view of its contribution to human well-being or should we, on the contrary, recognise that it has and non-instrumental value in itself? Awareness of the threats to biodiversity and the survival of the planet has not given rise to unambiguous ethical developments. Anthropocentric positions, emphasising the different types of values that nature may represent for humans, rub shoulders with biocentric ethical models, related to the saving of the certain specific species and recognising that the whole of the "biological community" has a value in itself, and ecocentric models, emphasising the intrinsic value of natural entities which extend this logic to things which do not belong to the biocenoses. What this diversity of approaches[1] shows us is that nature is at the same time the source of physical services and aesthetic pleasures and also of intrinsic and moral values.

Turner *et al.* (2003) chose a very broad typological approach to this question whose fundamental distinction concerns the anthropocentric character of the values.

- Anthropocentric values:
- instrumental anthropocentric value. It matches up with the economic notion of value and reuses the components of what will be defined as "total economic value": Use and non-use values (which will be specified below),
- intrinsic anthropocentric value. It expresses the value of non-human species "for themselves" but from a subjective point of view: humans define it according to their culture.

- Non anthropocentric values:
- instrumental non-anthropocentric value. It portrays the interest of entities for themselves and for the groups in which they are included (populations, communities, ecosystems),
- intrinsic non-anthropocentric value. This is an "objective" value. It expresses an entity's "inherent" value, independently of any "evaluator".

This list, which aims at exhaustiveness, seems *a priori* to extend beyond the scope of an economic evaluation, which must always be, *a priori*, anthropocentric: the only things which have value are those which are of interest to, have a utility for, human beings[2]. We will see the notion of total economic value (TEV), with its limitations and the controversies that it has provoked, is a practical attempt to shift this limitation.

The OECD manual (2002) chose a quite global view of value which reuses a large part of the typology above under three headings with vague limits:

- The instrumental values are derived from an objective function like the search for human well-being; as the classification of situations of well-being is based on preferences, the instrumental values of biodiversity are as well.

---

[1] In this report, we cannot go into all the aspects of a discussion in which many contributors, like S. Callicott, A. Leopold, A. Naess, B. Norton, P. Singer, M. Serre and many others, have intervened. The relevant cross-disciplinary analyses can be found in Larrère and Larrère (1998) or in the thesis of V. Maris (2008).

[2] We should note here that utilitarianism potentially includes in its calculation all of the beings capable of feeling pleasure and suffering. This fact opens the possibility of including animals with feelings, as proposed by the utilitarian philosopher Peter Singer in his work *Animal Liberation* when opposing the "speciesism", meaning discrimination based on the species, of the idea that only human interests deserve taking into consideration.

- The aesthetic values are obviously based on preferences but are not instrumental values, if we consider "beauty" to be an end in itself[1]; the diversity of landscapes or certain elements of biodiversity (including agro-biodiversity) is the support of the aesthetic values.
- The moral values are clearly not instrumental; they express the idea that biodiversity is the support of intrinsic or inherent values. The question of knowing whether the intrinsic values reside in the object or in the mind of the evaluator is an old and inconclusive philosophical debate.

The debate about the nature of intrinsic values will not be resolved here but it can help us to reconsider the question of well-being. Economic value has been defined as an instrumental value because it is directly linked to the contribution to well-being. As it is a question of human well-being it seems clear that this definition is sufficient to classify economic values as being "anthropogenic". This anthropogenic character means that the values are given by humans; but it may convey an interest in the support of the value itself, independently of its possible contribution to the objectives of the person who attributed the value to it. In this light, intrinsic values would be assimilated into the aesthetic preferences of human beings[2].

### *b. Is biodiversity substitutable?*

Staying within an anthropocentric perspective, let us go back to the question of the ability of agents to attribute values to biodiversity and ecosystem services that are commensurate with the values attributed to other goods and services. Essentially, to judge this ability is to answer the question: is biodiversity "substitutable"?

For the evaluation, the question is posed *a priori* from a subjective point of view: can the agents accept forms of compensation for losses of biodiversity? There may be several degrees of response to this question, depending on whether that means:

- obtaining a compensation "in nature", as proposed more or less *in fine* by the "*mitigation banks*" (Géniaux, 2002),
- obtaining an equivalent end service by different means; which raises the question of technological possibilities and judgement by the agents concerned of the quality of substitution[3],
- maintaining the psychological level of well-being by obtaining other sources of satisfaction (no doubt it is a more artificial context) as a replacement for the ecosystem services destroyed.

Therefore, the notion of substitution must be defined and, if we agree to use its widest sense, the limits of substitutability seem to be pushed back quite far. However, this way of thinking assumes that we consider that there is no real irreversible loss in the

[1] Here we come back to the "amenities of nature", whose possible disappearance J. S. Mill deplored.
[2] Callicott (1986) thus defends the idea that the intrinsic value of species is not independent of humans because it is conditioned by moral (human) values. There is no value without an evaluator.
[3] In particular, the substitution may appear imperfect, since the natural assets provide several services, particularly if certain services include amenities. Also, note that the history of techniques reflects in part the long march to produce artificially and in a way that is more controlled or better adapted to our needs, services offered by the ecosystems, starting with the invention of agriculture.

economic sense. But the natural assets destroyed may have symbolic values or a potential that it is difficult to completely explain and therefore to compensate for by the production of equivalents. The thesis defended by Arrow and Fisher (1974) was already that the losses caused by the building of a dam are really irreversible. Two categories of reasons may thus lead to the disqualification of the evaluation.

Firstly, the infinite or unacceptable character of the losses incurred. If certain choice options menace the survival of the human species, it is obviously difficult to measure variations in well-being. The unacceptability may be ethical: the species is not in danger, but it is its "authentically human" character – to plagiarize H. Jonas – that will be threatened. Destroying all of the species of birds or pollinators may not threaten the survival of man but certainly that of his "humanity". In both cases, between which the limits are no doubt fuzzy, economic evaluation would lose its pertinence or, more accurately, we would have to dispense with the cost/benefits perspective (how do we obtain the maximum well-being?) and adopt a cost/effectiveness perspective (how do we preserve the human species or its "humanity" in the most effective way?).

Secondly, the practical difficulty of obtaining a reliable result, i.e. a measurement that is both repeatable and sufficiently accurate to enable us to discriminate between the different choice options. Without going into the questions of methods here (see below), there are extremely scattered results for the measurements of values that should be attributed to most ecosystem services in the economic literature. Generally, the scattering increases when the evaluation moves on from a quantifiable service with a good degree of certainty, like the carbon storage or sequestration functions, to services that involve random processes, like bio-prospection activities.

Finally, with the specific features of biodiversity, we come back to the more general fact that economic evaluation *a priori* leads to much more reliable and robust results for marginal variations (small in the region of existing situations) than when is it a question of structural or fundamental changes.

Substitutability is at the heart of one of the questions that have enlivened the discussions on sustainable development, generally summarised by the opposition between the "weak" and "strong" concepts of sustainability. The "weak" concept makes the assumption that the different forms of wealth (financial or manufactured capital, human capital and natural capital) are quite widely substitutable in their contribution to social well-being, whereas the "strong" concept asserts the need to maintain sufficient levels in each field. The terms of the debate were already clearly set out in the confrontation staged in 1994 by the *Environmental Values* review. W. Beckerman considered that the weak concept was logically redundant with a rigorous standard economic analysis that clearly treated the question of equity between generations and that the strong concept, because it lacked clear foundations, could lead to unacceptable recommendations. The reply from H. Daly proposed notion of "critical natural capital", the limit value of the biophysical base below which the forms of capital became complementary (below a certain threshold, a reduction in the ecosystem services causes a reduction in the productivity of economic capital).

The question of critical natural capital provoked many debates (see, for example, the contributions to the work edited by Faucheux and O'Connor, 1997; Neumayer, 1999) and has been the subject of a special number of the *Ecological Economics* review, in 2003, from which we can retain the attempts to clarify the concept and the proposal of an original analysis framework (Ekins *et al.*, 2003) which, however, poses as many questions (in particular, about defining the sustainability standard) as it provides answers. Nevertheless, these analyses force us to consider the fact that natural capital has certain specific features for economic analysis, the most evident of which is that a minimum level is necessary for maintaining our own life ("*life-support function*") (Daily, 1997; Dasgupta, 2001).

### *c. Is biodiversity an economic good?*

The question may seem formal but it will enable us to specify certain points related to its evaluation. Economic goods have *a priori* a set of properties which, when they are not verified, may modify the attitude that we have to the object in question. We will repeat here the questions posed by O. Godard (2005) and try to give the most general possible answers to them.

Can biodiversity be seen as useful? A minimum of diversity seems essential to enable us to obtain most of the services provided to us by ecosystems (see the preceding chapters). But the agents' perception of that dependency is in general imprecise and usually very incomplete. Nevertheless, disregarding this lack of familiarity or competence, "nature's balances"[1] are sometimes a projection of the human mind. There are few works which can be used to argue that all of current biodiversity is necessary to the biosphere's balances and, without going back to the old categorisations into useful and harmful species, certain behaviours whose rationality can be recognised in the evaluations[2] show that all of the forms of diversity are not equally wanted.

Is biodiversity rare? We have seen that the answer to this question is ambiguous and should be nuanced according to the contexts. But there is certainly a demand for conservation that goes against the current trends of degradation. Therefore, we come back to the notion of "relative rarity" which is at the heart of the economic theory of value. But it is clear that many concerns posited by the ecologists involve what Baumgärtner *et al.* (2005) have qualified as absolute rarity and argue in favour of an expansion of perspective that takes in certain aspects of merit goods. This takes us back to the question of the measurement of biodiversity (see the preceding chapter) that the evaluations have run up against for a long time. Economic analysis typically distinguishes (Polasky *et al.*, 2005) measurements based on relative abundance (*a priori* the species, in spite of the limits of this notion for measuring biodiversity) from those based on dissimilarity (Weitzman, 1992)[3]. Some recent work tends to follow the ecological theories and attempts rather to evaluate a functional diversity and the relationships between the resilience of ecosystems and their ability to provide services (Perrings, 1998; Walker *et al.*, 2004; Kinzig *et al.*, 2006).

Can we control it? We have seen (chapters II and III) that human activity is indisputably at the origin of the threats to diversity today; but nevertheless that does not mean that, to plagiarize Michel Serre (1987), we have "the mastery of our master". The behaviours and developments responsible for the degradation are not intentional,

---

[1] For a quite complete account see the little work by P.-H.Gouyon (2001).
[2] See, for example, Zhang *et al.*, 2008.
[3] A pedagogical summary of this question and its issues can be found in Aulong *et al.* (2005) who also demonstrate, on an axiomatic basis, the impossibility of a single measurement including all of these dimensions.

in the sense that generally their objective is not to destroy and, conversely, due to the complexity of the working and regulation of the biosphere, it is never certain that the actions taken to protect nature will be effective in preserving its diversity[1].

Can we delimit physical units? Here we refer back once again to the preceding chapter. It is clear that there are multiple interconnections (matter, energy, genetic information, etc.) between the ecosystems and between their compartments, which does not necessary mean that sufficiently resilient units cannot be delimited. But the analysis is confronted with a multiple delimitation between the legal part, the ecologically functional area and the areas influenced by the phenomena of disturbance, circulation and migrations. On the basis of four local studies, Kinzig *et al.* (2006) showed that the cascading effects resulting from the change of management or use of certain surface area units did not respect the limits. From an evaluation point of view, this question brings up two distinct problems. The first is knowing what the pertinent area to evaluate is: surface area destroyed, ecosystems damaged, habitats disturbed or species displaced. These different notions do not match up spatially and must *a priori* be the subject of a differentiated analysis, as Fischer *et al.* (2009) pointed out, because the prospect of choice implies that we must define and classify the service they provide for us. The second is knowing if, in the case of biodiversity, having a local measurement is sufficient or if it is a question of estimating the contribution of the area concerned to a wider or global diversity, in functional or heritage terms.

Can we count homogeneous units? We can look for an answer to this question in the current work on inventories and mapping whose most successful ventures are no doubt those carried out by the European Environment Agency (Weber, 2008). The definition of equivalence classes will remain more problematic as long as there is no universal measure of biodiversity (see above) and, in practice, the institutions that have to organise compensation have to make do with quite crude indicators (see Géniaux, 2002) and build notions of ecological equivalent "in the expert's view"[2].

---

[1] Some works have shown that more species disappear in the protected areas (see Gouyon, 1994); which no doubt conveys the fact that we have protected the most sensitive areas and that their monitoring enables us to be aware of changes more quickly; but also means it is difficult to oppose the pressures that threaten ecosystems.

[2] S. Thoyer and S. Said (2007) present a procedure that employs this type of evaluation for the allocation of agri-environmental measures in Australia. In 2001, the State of Victoria's Ministry of Agriculture set up a pilot experiment called *Bush Tender* in the north of Victoria followed by a second experiment, in 2003, in Gippesland, two regions in which 60% of the local flora, located on private agricultural land, is threatened with disappearance. These programmes used an auction mechanism to allocate three-year contracts for the conservation of biodiversity. With the help of an adviser, each volunteer farmer set up a programme of changes to agricultural practices aimed at conserving local biodiversity. Then, he was invited to submit his specifications and the amount of the compensatory payments he wish to receive for the implementation of this programme. These offers were submitted in sealed envelopes to the decision maker who selected those that offered the best environmental gain for the least cost, measured using a biodiversity improvement score (*Biodiversity Benefit Index, BBI*). The *BBI* depended on three things. The first two were established by ecologists: the *Habitat Services Score (HSS)*, which measured the impact and the scope of this specification proposed by the farmer in terms of conservation of biodiversity, was revealed to the farmers to guide them; the *Biodiversity Significance Score (BSS)*, which measured the potential and importance of the site proposed for conservation in terms of biodiversity, remained unknown to the farmers in order to limit opportunist behaviours. The third is the amount *P* of the compensation demanded by the farmer for implementing his programme. Thus, the higher the value of $BBI = (BSS*HSS)/P$, the higher the probability of being selected.

Currently, it seems more acceptable to build consensuses about more "local" equivalence classes covering spaces that have multidimensional similarities.

What insertion into the agents' production or utility functions besides other substitutable or complementary resources? We discussed this point earlier. Here, we can point out the risk that choices based on agents' preferences may introduce a bias in favour of "charismatic" species or environments whose celebrity makes up for the agents' lack of familiarity (Moye, 1998; Metrick and Weitzman, 1998). This is like a characteristic of merit goods in that the preferences expressed do not reflect the well understood interests of the agents.

We will repeat O. Godard's (2005) conclusion accorded to which biodiversity seems to be an emergent property of other things considered to be resources: ecosystems from which economic agents obtain multiple services. Therefore, it is rather the ecosystem services that should be considered to be economic goods, in so far as we can describe and count them and measure more or less marginal variations. Biodiversity seems to be a parameter whose correlation with the intensity of the services seems to be validated in certain situations studied (see chapter IV). Therefore, when it is a question of the "value of biodiversity", we are mainly talking about the measurement of the impact of the variation in the biodiversity parameter on the social value of the ecosystem services.

According to Loreau *et al.* (2001, 2002) and Tilman *et al.* (2005), two main relations describe the connection between biodiversity and ecosystem services[1]:

- a positive correlation between biodiversity and the average level of ecosystem services,
- a positive correlation between biodiversity and the stability of those services[2].

Are the ecosystem services private or public goods? The answer is obviously that we must specify which service we are talking about. Some are private and even market goods, such as the hiring of hives to pollinate orchards or the gene banks used by seed companies, sometimes "trans-appropriative" like migrating birds can be. Others are common goods, like most sea fish stock resources or the landscapes of a national park. Most are mixed, meaning that they combine in the value of the same biophysical asset private dimensions, like the ownership of land, with public aspects, like the landscapes and, more generally, the non-use values. We should, in addition, specify the notion of "public":

- according to a spatial dimension, even though not very "excludable", the benefits of a landscape only concern those who can see it,
- according to a temporal dimension, the reserves of species and cultivars and the gene bank certainly have a current value, but they are also a form of insurance for future generations[1].

---

[1] Several works emphasise the fact that this relationship is not simple and Costanza and Fisher (2007) have proposed an analysis setting out several gradations from which the relationship can take different forms.
[2] This second relationship has been the subject of a long debate (see McCann, 2000); it tends to consider biodiversity as an insurance against negative productivity shocks on the ecosystems (Loreau *et al.*, 2003).

Therefore, the ecosystem services are only economics goods under certain conditions that are only realised in certain situations. Nevertheless, we should emphasise that this property is only essential for decentralising their management, as private goods managed by exchange or as public goods regulated by public incentives. It is not *a priori* necessary for the evaluation. In economic literature, ecosystem services are most often considered to be mixed public goods.

### *d. The problem with mixed public goods*

Mixed public goods are confronted by the same problems as "pure" public goods: the insufficient financing of their production by the mechanisms resulting from the rationality of agents and their expression on the "markets". Due to their link with the market, there is an added ambiguity linked to their private goods dimension.

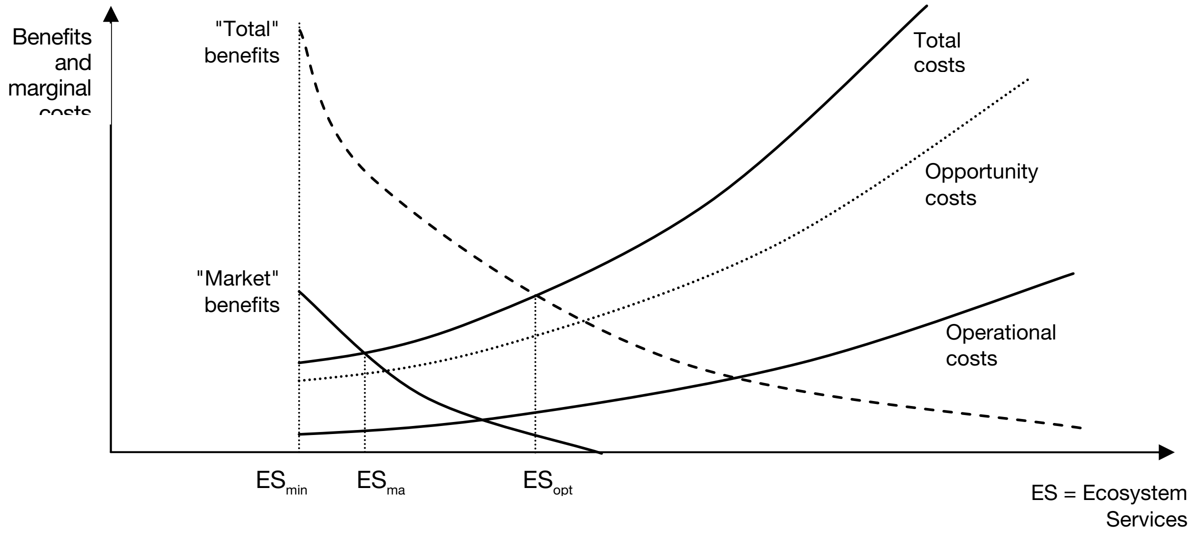
At this level, we must introduce a distinction between two types of public goods: those whose production can be effectively carried out in a centralised way (the public collectivity must then chose to get the goods produced by its services or delegate that task to private agent by contract) and those that are more effectively produced in a decentralised way (the collectivity must then set up mechanisms that incite private agents to contribute to the production of the collective goods). Whether goods belong to one category or the other is contingent upon characteristics and technical possibilities, on the one hand, and forms of legal and social organisation, on the other.

It seems that the production of ecosystem services and biodiversity should mainly be classified in the second and therefore the main problem posed to public authorities is that of regulation and financial and legal incentives. In a more restrictive view of biodiversity, which is simply a question of a collection of species or genes, we could consider conserving biodiversity by setting up as complete collections as possible and keeping them with a high level of security. This is what the seed companies do for an already constituted set of plant species of economic interest. No doubt this model could be expanded to areas of biodiversity of less recognised interest, within the context of public actions or services. But it does not seem realistic to think about expanding it to all of the elements that make up biodiversity and its dynamics which remained related to the conservation of areas which, particularly in Europe and in most advanced economies, have been privately appropriated.

---

[1] As shown, for example, by the Norwegian initiative of storing copies of certain seed collections at Spitzberg.

Figure V-4: The costs and benefits of the conservation of biodiversity




*Source: Pearce, 2007*

Figure V-4 gives a stylised representation of the situation, showing the difference that exists between the balance resulting from the confrontation, not necessarily on the markets but in the context of regulation institutions, between the directly perceived costs and benefits and the optimal situation that takes into account all of the benefits.

Point $SE_{min}$ symbolises the idea that there is a minimum quantity of services below which the balance and working of large-scale regulatory mechanisms would be threatened. We get back to the idea that the cost/benefits analysis would lose its pertinence for such situations. This point of view is largely consensual, but we should emphasise, along with D. Pearce (2007) and many other analysts, the crucial character of problems and information, one consequence of which is that it is very difficult in practice to locate the concrete conditions on this stylised curve. Therefore, we do not know at what distance from this limit we find ourselves.

We distinguish the "market" benefits linked to ecosystem services and biodiversity from the total benefits which are related not only to the public goods dimension but also to the imperfections of the appropriation and, more generally, control mechanisms. On the "costs" side, we separate the operational costs, directly linked to the financing of the conservation actions, from the opportunity costs, resulting from the fact that these actions involve, or will involve, because we can assume that many the actions of this type have not yet been undertaken, foregoing the use of the corresponding areas or resources for the production of useful goods and services (but of course having a lesser contribution to social well-being).

It thus clearly appears that the status of mixed public goods also leads to under production $(ES_{ma} > ES_{opt})$ and that socially optimal production involves the integration of the total benefits related to the ecosystem services into the choices. To do this, we must have a measurement of the non-market benefits whose social value cannot be directly observed and which are the subject of the economic evaluation. Reciprocally, the real economic cost of conservation includes *a priori* other things and expenditures explicitly agreed and must, in particular, include the opportunity cost of the means being mobilised for this objective. Insofar as the conservation decisions that go beyond the market equilibrium, most of them in fact, are taken most often by a public authority, we might consider that these choices take into account, as a priority, the costs that directly or indirectly affect that authority, but which may not be easily observable.

This fact leads us to point out a final difficulty, which is that the benefits related to the conservation of biodiversity may be mainly expressed at very different levels of social integration: local, regional, national, international or global. Biodiversity is thus qualified as a "multilayer" public good (in the same way as other resources that are common goods) to indicate that its total value depends *a priori* on the level of aggregation.

## 2.2. The notion of total economic value

The idea that nature is a source of amenities, i.e. of direct utility, in addition to productive resources, it is very old in economic analysis. Within the framework of welfarist theory, V. K. Smith (1987) defined everything from which agents can derive a utility (whatever the motivations) and which is rare as of economic value. A conceptual framework has gradually been elaborated so that it can take into account, more or less easily, all of the reasons for which humans see, or should see, interests in nature, ecosystems and biodiversity, in an inclusive framework.

In the case of biodiversity and ecosystem services, the importance of enlarging the evaluation framework was emphasised very early, particularly in the *Global Biodiversity Assessment* (Perrings, 1995) which showed that giving values to biodiversity for its direct use values alone was not sufficient, with a few rare exceptions, for justifying conservation actions in the face of the benefits of converting land. This analysis has been confirmed since then, for example, in the work of Simpson *et al.* (1996) on the benefits hoped for from bioprospection.

The distinction between market values and non-market values has already been introduced but we should emphasise that the fact that a service seems to be the subject of real transactions does not imply that the prices reflect the marginal utilities. The match is only assured if the rights are correctly defined and respected, if there are no significant residual externalities, if there is no "market power" (monopolies or monopsonies, in particular) and if there are no problems of information. We can simply illustrate this sometimes very complex problem by remarking that a large part of biodiversity or the ecosystems at the origin of multiple services is located on private land whose appropriation is clear: which does not, evidently, resolve all of the questions posed by the social management of those assets.

The evaluation of biodiversity is sometimes presented as the cost of inaction (Heal, 2005; Braat and ten Brink, 2008). Without under estimating the pedagogical virtues of a well chosen semantics, the cost of inaction is nothing other than the value of what one is losing, due to the absence of appropriate political actions[1] and by not changing our individual and collective behaviours. We can only think that presenting the benefits of conservation as the costs of non-conservation has the effect of reinforcing the psychological perception of the issues on the part of those we are addressing (this is

[1] The acknowledgement of the lack of appropriate policies does not imply any judgement about the question of their justification or timeliness. Nor does it exclude one.

one of the divergences between CTR and WTP). It is also a way demonstrating that, from an economic point of view, an asset's social value cannot *a priori* exceed its replacement cost[1].

We are going to review the conceptual foundations of the value of ecosystems and biodiversity by explaining the main categories chosen by the economic literature. The list and the boundaries of these categories sometimes vary, depending on the authors or the situation. The presentation below aims for clarity and consistency, but there are other possible choices.

The total economic value (TEV) of ecosystems (Pearce, 1993; Pearce and Moran, 1994; Turner, 1999) is traditionally broken down between:

- use values which include the benefits derived by the agent from the consumption of the assets and the practices linked to the assets but not leading to their consumption ("*consumptive and non-consumptive uses*"),
- non-use values which convey the benefits derived by others in cases where the agent's utility function includes ethical or altruistic preferences.

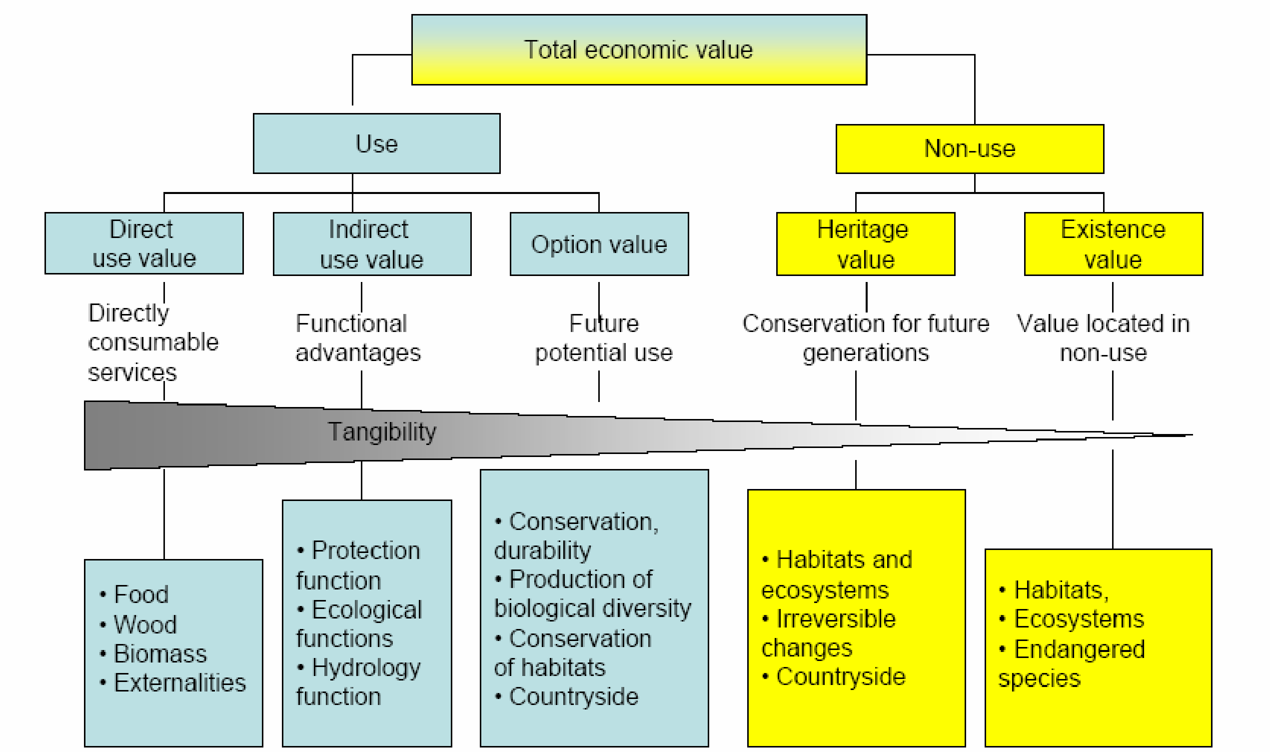
The construction of a total economic value implies a form of aggregation of these different values. Perman *et al.* (2003) stressed that there are several formulations, depending on whether the focus is on the environmental costs (EC) of the projects or, symmetrically, on the total value (TV) of the assets threatened:

EC = use values + existence values + option value + quasi-option value
TV = use values (direct and indirect) + non-use values (option, existence...)

Not only does the lack of uniform terminology reflect the existence of different approaches to the same question, it also shows up the lack of precision of a theoretical corpus that is still debating the list and limits of the different subcategories. In addition, we can remark that the difficulties of measurement limit the practical issues of standardisation.

[1] The question then is to ensure that the replacement possibilities effectively cover all of the values that the assets supports.

Figure V-5: The values of ecosystem services




*Source : Strategic analyse centre, February 2008*

Can we simply sum the components of the total economic value? The question may seem superfluous, as the TEV has been specifically developed to obtain a wider value than the direct use values alone. However, the notion of total value is still controversial for a set of conceptual and practical reasons.

The fundamental reticence appears to be linked to certain ambiguities: the difficulties of delimiting the issues and limitations of anthropocentrism, possible confusion between an intrinsic value of nature, independent of humans, and measurements of surplus variations related to "marginal" losses of natural assets, difference in the natures of components, some of which are very far from a consumption choice option and reticence to about summing benefits and costs without reference to their social distribution.

Added to these concerns, are reasons related to more technical difficulties: the risks of duplicate counting or the simultaneous valuing of socially and technically incompatible uses or services, the eventual difficulty of choosing a reference situation (A world without humans? The situation which prevailed one or more generations earlier?), the lack of robustness of measurements in time and space (and even between two samples) and, no doubt, the fact that errors or blunders have destroyed the credibility of some existing evaluations.

Fromm (2000) also noted that the items be taken into account in the total economic value tend to multiply (contribution value, inherent value, indirect value, infrastructure value, etc.) and that these categories refer to distinct levels of biodiversity and, above all, to complementarity relationships that are specific between species or between certain parts of structures or ecological functions and human well-being. The existence of these relationships has multiple implications on the changes to the TEV when the assets change and therefore consequences on the way of taking them into account in the decisions.

Nevertheless, the different components of the TEV, as well as the difficulties that they may raise, need to be specified because practical experience of study and research work has shown up a less simple reality than the one that figure V-5 summarises.

## 2.3. The certain use values: biodiversity as a source of goods and services

Traditionally, we distinguish between the direct uses, for which behaviours are observable and the sometimes market services (like ecotourism, but indirectly, recreational hunting imply large market budgets), and the indirect uses which correspond to the benefits derived from the regulation and support functions carried out by more or less biodiverse ecosystems and from which the agents benefit without being in interaction with them and, often, without being clearly conscious of them … as long as these services are not threatened or destroyed[1].

### *a. The direct use values*

They correspond, without the boundaries always being well defined, to the three main groups:

- the direct consumption uses, for food, biomass energy and medicinal plants,
- the productive uses as industrial resources (including pharmaceutical), sources of energy or building materials,
- the uses not implying consumption, like recreational or aesthetic uses, tourism, sciences and education.

The main distinction here is between a goods and services that "consume" the ecosystems or modify their diversity and those that derives benefits from them without affecting them. The boundary is not as simple as the list could lead us to believe because a reasonable harvesting of fruit or wood may enable the ecosystem to maintain itself sustainably in quality and quantity whereas a too intense recreational use may lead to its impoverishment, degradation or destruction.

### *b. The indirect use values*

These correspond to the benefits related to the maintaining of ecosystems that provide services that do not imply direct interaction, such as:

- services contributing to the productivity of agrosystems,
- local regulation of climates,
- maintaining the fertility of soils,
- controlling run-off and hydraulic flows,

[1] Green *et al.* (1994) distinguish between primary and secondary values for a quite similar typology.

- purifying water or the atmosphere,
- sequestering and storing carbon, etc.

The fact that there is a relationship between ecosystems' diversity and their productivity has been shown by many studies (Tilman, 1999; Tilman *et al.*, 2005). Economic analyses have demonstrated that a greater diversity increases the probability of containing individuals or species able to profit from circumstances and that greater diversity is linked with a lower variance in the productivity of ecosystems ("portfolio effect"). This argument assumes an independence of the species present (biodiversity as a library) whereas, on the contrary, the species present in a same ecosystem are very interdependent, but this interdependence also reflects the existence of complementarity that may accentuate this effect.

The second part of the INRA study on the relationships between agriculture and biodiversity (Leroux *et al.,* 2009) listed and analysed the different ways in which greater diversity of cultivated or wild ecosystems could contribute to producing services for agriculture. The exercise seemed to be tricky because there are not many works that clearly show a clear positive relationship between biodiversity and agricultural productivity and a significant part of the modernisation of agriculture after the Second World War has specifically consisted of freeing agriculture from these services, which also included "disservices", as we have been recently reminded (Zhang *et al.*, 2007).

We should stress the fact that these values do not only or necessarily correspond to current effective uses. They also concerned future uses. Therefore, the use values that should be retained for the evaluation of ecosystem services correspond to values discounted over the whole period considered, which can be indefinite. A question of great practical importance concerns the existence of uncertainties[1] about future uses (as is generally the case). The use values can integrate this uncertainty by defining a probability distribution on the different scenarios, for example[2]. However, the existence of uncertainty has specific effects on the comparison of the alternative options.

## 2.4. Use values in uncertainty: biodiversity as insurance

The uncertainty which frequently affects the availability of natural assets may modify the way of understanding the benefits derived by the agents. The literature on cost/benefits analysis shows that, in the presence of on uncertainty, several concepts could measure the variation in the benefits that the agents derive. What is the correct measurement of use values in uncertainty? Meier and Randall (1991) showed that the choice depended on the way that the institutional context allowed the agents to use insurance and compensation mechanisms. This is a quite technical question and here it will probably be sufficient to retain that, if there is an equitable insurance system, then the hope of surplus or the option price may be used. However, there are

[1] Economists distinguish the "risky" situations, on which they can define probabilities, from the "uncertain" situations for which they cannot be attributed. The word "uncertainty" is used here in the generic sense of lack of certainty.
[2] The agents' risk aversion can even be taken into account by calculating certain equivalents that under evaluate the scenarios that lead to an increase of the risks.

situations that are uninsurable in the usual sense because the current choices modify the space of future choices.

### *a. Maintaining the future choice options*

Independently of current or future use, an asset, whether natural or not[1], can have an **option value** which corresponds to an added value related to the possibility of not exercising the option. This concept, which is current in finance today, was proposed by Weisbrod (1964) as an argument in favour of public financing of the protection of natural areas in the United States. The option value was the subject of intense debates in the 1970s, one of the sensitive points being the understanding of the conditions which determine whether its sign is positive or negative. Option values must not be confused with uncertain future use values, even weighted by a risk aversion factor.

Today, we distinguish between two options values, depending on whether the uncertainty concerns the future behaviour (the decision maker does not know, at the present time, whether he will consume the goods) or the utility that will be derived effectively from its use in the context of increasing information and choices between more or less reversible options (Arrow and Fisher, 1974; Henry, 1974). The second case (quasi-option or dynamic option value) is the more interesting here because it corresponds to situations in which it is a question of deciding on the destruction of a habitat or the increase of threats to a species (probability of extinction) in a context where the possibilities of future uses are imperfectly but better and better known (it is sometimes interpreted as being a measurement of the value of the anticipated improvement in the information. Hanemann,1989).

However, this interest should be seen as relative because in order to measure a quasi-option value we would have to have a complete description of the scenarios linked to each option and an evaluation of all of their consequences[2]. This requirement is obviously unrealistic and, in practice, the quasi-option value can only be calculated in the context of restrictive hypotheses. Bosetti *et al.* (2004) studied the question of the allocation of land and investment choices related to partly degraded land between (1) remediation and return to a more natural state (2) irreversible development. They were able to calculate the option value to be attributed to the flexible option in a concrete situation related to the city of Ginostra on the Isle of Stromboli (Italy).

Kassar and Lasserre (2004) formalised biodiversity in a framework of real options where the resources are substitutable (financial portfolio approach). How do we optimise the conservation decisions when every loss of biodiversity is irreversible and

---

[1] Today, the theory of real options is at the base of the evaluation of financial assets and the economic analysis of investment choices.

[2] C. Henry particularly stressed this limitation during the conference by P. Sukhdev in Paris (IDDRI-EHESS, 25th of November 2008). An illustration of this type of issue in the case of biodiversity was proposed by the film by Hubert Sauper, *Darwin's nightmare*, on the consequences of the introduction of Nile perch into Lake Victoria. For a wider perspective, read the work by J. Diamond (2000) on the origin of inequalities between societies that shows how, at the dawn of the Neolithic, the possibility of domesticating local areas was a determining factor in the transition to agriculture which, by creating the possibility of feeding dense populations and producing a food surplus, allowing for the division of labour, played a determining role in the development of societies and their tests of strength and therefore in the history of humanity.

the future values are uncertain? They showed that substitutability, normally considered as reducing the value of a species, is in fact a source of value[1]. However, this type of approach can be criticised because it presents biodiversity as being a collection of independent species whereas modern representations insist rather on the importance of interactions.

### *b. Diversity as insurance*

New human diseases or crop pests may appear or become widespread and greater biodiversity increases the chances of finding the molecule or mechanism that will enable us to defend ourselves more effectively. In this sense, the option value of biodiversity may be interpreted as an insurance premium that the agents agree to pay in order to reduce the possible consequences of the occurrence of potential risks (Perrings, 1995b). Loreau *et al.* (2003) have shown this insurance function at the spatial level.

Baumgärtner and Quaas (2005) distinguished between the public and private insurance value of the conservation of biodiversity, because the incentives affecting the managers (particularly agricultural) are directly affected by it. Baumgärtner (2007) proposed a measurement of the insurance value of biodiversity for the production of ecosystem services. Within the framework of an ecological economics model, Baumgärtner and Quaas (2007) showed that maintaining greater agro-biodiversity was equivalent to an insurance mechanism.

## 2.5. No-use values: biodiversity as heritage

The interpretation of existence values has changed significantly since they were introduced by J. V. Krutilla (1967). Originally, it was a question of taking a willingness-to-pay (or to forego benefits) to maintain the existence of certain natural assets into account in the utility of agents without concern for present, future or potential use (hence the term "passive use"). The dimension of altruism towards the nonhuman species or nature in general then emerged, based on ethical or religious motivations. We can see in this change the idea that agents take on (in their motivations or well-being) the ethical values that it did not seem could be integrated into an anthropocentric framework. This idea comes up again more or less clearly in the notion of "*stewardship*", i.e. in an ethical system in which the subject realises that it is a Man who names, and that *in fine*, he cannot shed his responsibility. Therefore, it is essential to make sure that the motivations which underlie them, whatever their nature, will be effectively taken into account in the collective choices.

[1] The marginal value is decreasing in the number of species but increasing and convex in the value of the marginal species. For a homogeneous model with two species, they show that volatility is a factor in increasing the value of biodiversity, whereas correlation reduces both the total value and the marginal value of species.

### *a. Altruistic values*

Amongst the non-use or passive use values, the literature finally distinguishes three forms of altruism[1] or, rather, subjects on which it is exercised:

- altruism towards our contemporaries, which makes us value the conservation of ecosystems so that others can benefit from them; this is the notion of "*vicarious use value*"
- altruism towards our descendants or, more generally, future generations to whom we hope to leave functional and usable ecosystems "*bequest value*",
- altruism towards nonhuman species that we may recognise have a certain form of moral right to exist ("*existence value*" in the strict sense, sometimes confused with "intrinsic value" which is the product of a non anthropocentric perspective[2]).

Here, we will stay within a perspective where the agents have preferences that condition their well-being, but these preferences are more complex and include ethical considerations. However, this approach encounters a practical difficulty in aggregating non-use values: which reference population? In their study of the value of Scottish and British forests, Willis *et al.* (2003) ran into this difficulty when mixing potential use values and more altruistic values. They arrived, not without humour, at the conclusion that if you decide (arbitrarily) to sum these values without effective use over a whole national population, British forests are automatically attributed a much higher value than Scottish ones due to the sole fact that the difference in the population of these two political entities (and undoubtedly in a way that is not very sensitive to the real interest that these two populations show in their "national forests").

Nevertheless, the non-use values can have a predominant importance in the willingnesses-to-pay (WTP) recorded. Several studies have explicitly dealt with the question. Stevens *et al.* (1991) asked the people questioned to divide their WTPs between the different types of value. As answers, they obtained: 7% for the use and options values, 44% for the bequest value and 48% for the existence value. McConnell (1997) repeated the results of a contingent evaluation study carried out in 1993 on the porpoise which showed that a significant number of individuals are altruistic: their willingness-to-pay is higher if they know that other people could benefit from observing porpoises.

### *b. Is civic involvement an economic value?*

In a controversial article, Kahneman and Knetsch (1992) argued that individuals were simply looking for moral satisfaction by declaring WTPs for the non-use values. This result was contested, notably by Smith (1992) who countered that the economists did not have access to the "true value" of assets and that the validity of a result could not

---

[1] This utility could convey the satisfaction, for ethical, religious or spiritual reasons, of leaving a richer biodiversity to future generations or for nonhuman species (Barbier *et al.*, 2008).

[2] From a bio or ecocentric perspective, the intrinsic values cannot be the subject of an evaluation. They may underlie an "all or nothing" way of thinking that is not very pertinent for guiding practical choices.

be discussed unless a significantly different result had been obtained by another means. However, the idea that non-use values are not economic values is found again in the Opaluch and Grigalunas (1992) who talk about ethical and loyalty choices, Diamond and Hausman (1993) on charitable gifts, and Common *et al.* (1997) and Sagoff (2004) on civic behaviour. Therefore, there is a whole current of literature that calls into question the economic nature of non-use values (or, at least, the measurements made of it and which are not consistent with the economic theory of value), even leading some authors[1] to suggest that they should not be taken into account in evaluations or, at least, they should be taken into account with discernment (see Arrow *et al.*, 1993).

Recently, Spash *et al.* (2009) analysed the reasons underlying the willingness-to-pay for the improvement of the biodiversity in a hydro system. This study shows that the classical economic variables are much less explanatory than the socio-psychological or philosophical variables of the answers obtained in a contingent evaluation. Their conclusion is that alternative approaches to the measurement of these many level values must be implemented to assess the validity and significance of the results. The authors are referring here to the mixed approaches that afford a large place to the deliberative dimension.

* * *

**Therefore, the taking into account of a valuation of biodiversity and ecosystem services wider than direct use values alone seems to be a determining factor but must be the subject of a differentiated assessment:**

- The indirect uses make up a large part of the ecosystem services. The absence of direct interaction implies specific approaches, but it seems necessary and possible to take them into account.
- The option values may form an important motivation for conserving nature but we must expect their measurement to be imprecise, contingent on restrictive hypotheses and, no doubt, under evaluated.
- The non use values are not the subject of consensus among economists, a consequence of which may be that taking them into account, though sometimes necessary and legitimate, cannot be entirely based on economic methods and therefore implies recourse to specific treatment methods.

We must now discuss the temporal dimension of the evaluation and in particular the questions of discounting and uncertainty.

## 2.6. Taking time and uncertainty into account

Not all of the consequences of choices made at a given time are immediate. In the matter of the environment and in particular ecosystem services, some effects may

---

[1] For these authors, utility must be based on egotistical motivations alone. Thus, Diamond and Hausman (1993) consider that "*benefit-cost analyses and compensatory damage assessments should not take into account ethical values: instead they should be based only on self-interested (economic) preferences*".

even be very far in the future. This evident fact has a dual consequence on the evaluation:

- on the one hand, the costs and benefits related to the far future consequences tend to be given less weight in the decisions,
- on the other, the future occurrence of consequences may be uncertain, bearers of potential surprises or controversial.

### *a. The choice of a discounting factor*

Discounting is a method used to adjust flows of values that are not directly comparable because they occur at different dates on the same basis. This enables us to compare projects which do not have the same temporal profile of costs and benefits on the basis of their *net present value*, which is the sum of the benefits and costs of each period adjusted to the time of the decision by multiplying them by a "discounting factor". This factor $F(t)$ may be considered to be the product of annual factors, generally expressed by an annual rate $ai$ or $a$ if we assume it to be constant[1]: $F(t) = \Pi[1/(1+\mathrm{ai})]$.

Discounting expresses *a priori* two phenomena: agents' preference for the present (impatience) and the expectation of a lesser future marginal utility of money related to the growth of incomes[2]. Though discounting reflects observable phenomena (the agents really prefer the present for most of their concrete choices, the banks only lend capital *a priori* in exchange for a positive real rate of interest[3], etc.), it is sometimes criticised on an ethical level because the undervaluing of the future issues tends to downgrade the long-term, which may lead to neglecting certain serious but far-off consequences. Therefore, discounting does not seem to be very compatible with the perspective of equity between generations.

Taking the preferences of future generations into account is a tricky question, conditioned by multiple hypotheses as to the state of the world (here, the ecosystems), technologies (destruction that today seems irreversible may no longer be in the future, essential resources might become substitutable … or the contrary) and ways of living (to what will our descendants attach most value?). An alternative, underlying many discussions on sustainable development, is that equitable treatment consists of preserving freedom of choice by avoiding the irreversible destruction of the alternative options.

Could we forego discounting? If we do not discount we will avoid favouring one period over others but it will make it more difficult for us to compare different options[4].

---

[1] Expressing discounting by an annual rate (which is of the size of a real interest rate, i.e. reduced by inflation which is the currency's loss of value) makes it easier to understand and makes the discussions about its choice and use easier.
[2] The choice of the public discounting rate (there are *a priori* as many private discounting rates as there are decision making centres; the real interest rate in a perfect financial market would reflect *a priori* the social discounting rate) reflects a forecast of the future economic growth.
[3] The real rate is obtained by subtracting the rate of inflation from the nominal rate.
[4] For example, when the benefits are not limited in time, the total utilities of all of the scenarios become infinite, as do the values of durable goods, like the ecosystem services which should then be conserved "at any cost"… But several incompatible objectives would thus be attributed infinite values whose comparison becomes more difficult.

However, there is a whole literature on this question, some elements of which we will mention:

- the "golden rule" of growth theories (Solow, 1966) is to prioritise the option that maximises the level of utility in the very long term,
- the long-term dominance criteria ("*overtaking*") is a variant that consists of choosing the option that enables you to maximise the sum of utilities after any date later than a date T,
- the *maximin* criterion  (Rawls' criterion) specifies the prioritising of the option which provides the maximum well-being for the least well-off generation,
- Chichilnisky's criterion (1996) expresses a sustainability imperative defined by the respect of certain axioms (in particular, avoiding the dictatorship of the present and the dictatorship of the future) which leads to maximising a convex combination of the discounted utility and the very long term level of utility.

These different approaches are very contingent on the representation of developments in the very long term and they do not yet have practical expressions as flexible as those attached to discounting. In practice, the long-term may be better taken into account by choosing a very low discounting rate[1].

The *Stern Review* on climate change chose a very low "impatience" (0.1% per year), a low risk aversion (= 1), a growth in world consumption per head of 1.3%, giving a discounting rate of 1.4% per year, significantly lower than the recommendations of all of the reference works. N. Stern justified his choice as being "prescriptive (= merit)" and not "descriptive (real long-term interest rate of the markets). If inequalities were taken into account, it would be lower still (Atkinson's theorem). These choices been the subject of contradictory judgements, to which we refer the reader (Godard, 2008; Heal, 2008; Weitzmann, 2008).

For the very long term, economists of the environment tend to prioritise "hyperbolic discounting" (the annual discounting rate decreases in inverse relation to the distance of the temporal horizon) even though this practice can lead to temporal inconsistencies[2]. Several empirical studies have shown that this technique is a quite good reflection of the reasoning implicitly used by the agents (see, for example, the work edited by Portney and Weyant, 1999, particularly the contributions of K. J. Arrow and M. Weitzman).

This principle was used in the Lebègue report on the "price of time" which recommends a rate of 4% per year for the first 30 years the its gradual reduction according to the formula $r_t = (1{,}04^{30} \,.\, 1{,}02^{t-30})^{1/t} - 1$ after the 30th year, to reach about 3% at the end of 60 years and converge to 2% in the very long term[3].

---

[1] Note that the real long term interest rates of financial institutions, which are compared inevitably to the discounting rate, even though they are not really equivalent, are frequently very low and especially in the current period of financial crisis.

[2] The classification of two choice options with different temporal profiles may change when we move in time because the value of the discounting factors of each period varies.

[3] Weitzman (1999) suggested comparable rates, but extended the reduction to 0% after 300 years, because he considered that the agents are indifferent to the date of occurrence of events for very distant time horizons. For the evaluation of road projects, by simplification, the infrastructure services use a rate of  4%, then 3.5% between 30 and 50 years after the start of work and 3% after 50 years.

Finally, we should stress that discounting concerns the utility (and not the material world). Therefore, it applies to prices or their equivalents resulting from the evaluation methods. In the case of a quite distant future, it is legitimate and even necessary to consider differentiated changes in relative prices. The prices of manufactured goods may drop due to technical progress and the prices of natural assets should rise (Krutilla and Fisher, 1975) due to them becoming rarer (and certain services related to biodiversity are almost certainly "luxury goods" whose income elasticity is greater than one[1]). But technical progress may also have the effect of making certain ecosystem services more easily substitutable.

In the report on *The shadow value of carbon* (CAS, 2008b), the relative price of carbon was considered to be increasing like the discounting rate (nuanced application of the "Hotelling rule"[2]) until 2050, which means that its discounted price at today's date (2008) is constant. After that, due to the exhaustible character of hydrocarbons, there is a period of decline in the relative price of carbon and then a period when it will be null. Therefore, the prospect is significantly different but draws our attention to the treatment we should give to the exhaustible character of biodiversity.

**We can retain the principle of applying the discounting factor generally used for public choices to the questions of biodiversity and ecosystem services, making sure that we choose the changes in relative prices in a transparent and pertinent manner. For example, a rate of 4% per year, applied to relative prices that are increasing by 1% per year, over the first 30 years, then decreasing according to the recommendations of the Lebègue report[3].**

That being the case, the main difficulty will then be linked to the representation of the future making sure that we describe a pertinent way the irreversible situations and losses of resilience.

### *b. Resilience and irreversible losses*

It is difficult to foresee the consequences of the destruction of an ecosystem or damage to it. The balance and development of ecosystems are dependent on multiple external and internal factors, whose interaction results in what are known as non-linear dynamics. The evaluation must make a choice, which amounts to either calculating a certain equivalent (including probabilistic measurements and maybe risk aversion, i.e. a reduction in the expected utility reflecting the fact that the agents prefer certain

---

[1] The question of income elasticity of the environment demand has been the subject of a long-standing debate whose under-lying idea has for a long time been to consider that the environment is a "luxury good" whose income elasticity of demand is greater than one (see Martinez-Alier, 1995). This question comes up again in the wider debate about whether there is a Kuznets curve for the environment. A recent empirical meta-analysis (Jacobsen and Hanley, 2009) validated the idea of a positive sensitivity to income of the willingness-to-pay for biodiversity, but with an elasticity of less than one.

[2] The "Hotelling rule", a result of the theoretical analysis of the pricing of exhaustible resources in an optimum growth model, specifies that their prices should change at the rate of the discounting rate starting from the equilibrium price, which amounts to using constant prices in an evaluation over time.

[3] In practice, by applying this rule, a discounted value over an infinite period is equal to the value of the services for the current year multiplied by 41,5.

results)[1], or opting for a forecast scenarios procedure representing each set of hypotheses (as in the *Millennium Assessment*).

The question of discounting, tricky though it is, must not mask the uncertainties about the development scenarios of biodiversity and ecosystem services (Polasky *et al.*, 1993; Perrings, 1995; Weitzman, 1998, etc.). Knowledge about the dynamics of ecosystems threatened by a project is generally neither complete nor perfect. In other words, evaluation of a project's impact on biodiversity and ecosystem services poses, on the one hand, conceptual problems related to the understanding of the ecological dynamics and their relationship with the services provided and, on the other, questions of information about the situation of ecosystems and the way in which they will be affected by the project. Therefore, the choice of the equivalent prices to be used to evaluate a project's consequences is contingent upon the pertinence of the hypotheses selected by the evaluator and quality of the information at his disposal.

Damage to ecosystems is frequently expressed by economists in terms of irreversible losses, interpreted as losses of future choice options and conveyed, as we have seen, by the addition of an option value. Tilman *et al.* (1994) introduced the notion of "extinction debt" defined as he future ecological costs of the destruction of a habitat, in terms of (delayed) losses of species diversity. Leroux *et al.* (2008) showed that taking the effect of an "extinction debt" into account led to the definition of an increased option value for the choices that avoid these effects.

This procedure seems to express the issues of an isolated decision well but we are not sure that it is sufficient for handling the consequences of a gradual reduction in biodiversity. In practice though we can envisage the prospect of the exhaustion of a resource such as petrol or coal[2], it does not seem possible to imagine the future of humanity without more or less diverse ecosystems. Therefore, it seems that we should break the question down into the three hypothetical situations that take us back to the discussion on the substitutable character of biodiversity:

- irreversible losses of technically substitutable elements of ecosystem services,
- losses of irreplaceable elements of biodiversity but the imaginable consequences of which do not threaten the survival of our societies,
- losses of indispensable elements of biodiversity, the unpredictable consequences of which threaten the survival of our societies as we know them or even the future of humanity.

The characterisation of these situations in reality raises many difficulties, but here we can discuss, the way that they should be treated in the context of economic evaluation. Apparently, the first case raises no specific difficulty, except for the specification of the changes in the relative prices of ecosystems that become rarer

[1] Though practice has sometimes legitimised the use of increased discounting rates to take account of the growth of risks (mixing discounting with Arrow-Pratt type risk aversion), modern discounting theories (see, in particular, the work of C. Gollier) tend on the contrary to consider that dominant effect of the increasing uncertainty about the increase in incomes is to limit the decrease in the marginal utility of the currency and should therefore by translated by a reduction of the discounting over time. This is one of the theoretical justifications of hyperbolic discounting.
[2] In so far as there are substitutes, the only consequence of exhausting these resources is to oblige us to imagine carbon free economic development.

and the possible loss of option values[1]. The second is more difficult and probably corresponds to the application conditions of the "Hotelling rule" (changes in prices at the rate of the discounting rate, implying a potentially considerable discounted value if the consequences of the loss continue to be felt indefinitely). The third case tends to disqualify economic evaluation from its cost/benefits perspective (as the costs are infinite, the losses must be avoided "at any price"); the cost/effectiveness perspective could be used to choose the least cost amongst the alternatives.

That having been said, in the "ordinary" cases in which economic value is a priority item determining choices we must make sure that the value taken into account reflects the issues correctly. C. Perrings (1995, 1998) pointed out that the main consequences related to a loss of biodiversity could lie in losses of resilience. But Hanley (1998) showed the difficulties or even the impossibility of taking the effects in terms of resilience into account in the context of an economic evaluation, because we cannot generally characterise them in a relevant way. Therefore, it is important that the decision maker makes sure of the quality of the information he or she has on the consequences of choices on the structure and working of ecosystems. M. Sagoff (2008) pointed out that in many cases ecological services involve indivisible factors that cannot easily be taken into account by an incremental analysis "at the margin": the economic value attributed to the units of certain services could well rightly be very low, because we are very far from a tipping point. Therefore, the notions of a "*threshold effect*" and the non-linearity of ecological systems (Brock and Xepapadeas, 2003) seemed to be determining factors for evaluating the effects of a choice which affects the structure or working of ecosystems and whose consequences also depend on the capacity to adapt of the populations concerned (Carpenter and Brock, 2008).

**The variability of ecosystem parameters in the context of cycles whose amplitude and regularity also vary within a statistical range, often not very predictably, and the existence of irreversible effects and threshold effects in the non-linear dynamics of the relationships between ecosystems and societies should lead us to examine particularly carefully the hypotheses made about the changes in the relative prices. Even though the hypothesis of a gradual increase seems acceptable as a first approximation for general biodiversity, multiple individual cases must be the subject of a specific examination.**

## 2.7. To conclude

Biodiversity and ecosystem services contribute to human well-being in many ways and there is no doubt therefore that humans attribute a value to them. Economic value does not exhaust all of the reasons that the philosophies may consider legitimate, but it is commensurate with the values attributed to other goods and services contributing well-being and, above all, with the opportunity costs of conservation or the technical costs of restoration or compensating in kind. On this account, it corresponds to what is expected from reference values for guiding choices about substitutable assets. If the questions concern higher order issues, involving ethical preferences or civic judgements (which may amount to considering biodiversity to be a merit good), the

[1] Setting a political objective, such as stabilising the level of biodiversity in 2010, creates a rarity reference. We could then use a lesser version of the Hotelling rule (inspired by Schubert, 2008) consisting of changing the prices at a rate equal to the discounting rate reduced by the rate of rebuilding/restoring the ecosystems destroyed.

arbitrations should no doubt include considerations of an equivalent order. For example, if an infrastructure project threatens a habitat enjoying legal protection, we cannot directly use a simple economic calculation as an argument against it. The classification of a species or an ecosystem is a compromise between heterogeneous interests and calling it into question no doubt implies that all of the interests concerned should be represented once again.

In the evaluations, the question of time and discounting should be treated according to the general recommendations in this area on condition that the hypotheses about the changes in relative prices are consistent with the underlying representation of the future and that the uncertain or controversial situations are the subject of explicit and socially acceptable treatment.

The complexity of certain questions raised by the evaluation of biodiversity and services provided by ecosystems leads us to stress that taking them into account in a socio-economic evaluation must involve as accurate an analysis as possible of the biophysical consequences of projects, especially in the spatial and temporal terms, which leaves a place to explicit uncertainties and legitimate controversies. Once this analysis has been done, the transition to economic values cannot be reduced to the assessing of expenditures. It must make sure that it respects the amplitude of the consequences of the particular changes on the well-being of the populations concerned, in the knowledge that these populations may not live nearby, due to the existence of indirect effects, or not be in physical interaction with the assets concerned but attribute non-use values to them. It is to meet this objective that a set of methods aiming to produce indicators in the form of prices have been developed.

## 3. The monetarisation methods

There is an abundant literature on the methods used to measure the value of natural assets, both from an *a priori* perspective, to take the environmental impact of projects into account, and from an *a posteriori* perspective, for indemnifying damages, especially those caused by accidental pollution, such as oil slicks (OECD, 2002). After a period during which these methods met with widespread distrust, linked both to controversies about the magnitudes actually measured and to doubts about the use of results whose robustness had not been proven (again recently, many authors have pointed out the extreme technical and conceptual difficulties of performing this exercise in the case of biodiversity: Hanley *et al.*, 1995; Nunes *et al.*, 2001; Hanley and Shogren, 2002, etc.), today, the pertinence and the weaknesses of each approach are better known. Their practical conditions of use are even sometimes codified, especially in countries that have made cost/benefits analyses (CBA) a mandatory prior condition to any expenditure of public monies[1].

The methods that will be presented aim *a priori* to construct indicators with the price dimension and that reflect the agents' willingnesses-to-pay (WTP) and, by aggregation, those of the populations concerned, to maintain the benefits linked to the

[1] In particular in the United States, where the Carter, Reagan and Clinton administrations successively promoted the development of CBA for choices related to the environment and protection by issuing *Executive Orders* 12044, 12291 and 12866. Currently, American law imposes the evaluation by a CBA of any regulation policy whose impacts are significant.

uses and existence of biodiverse ecosystems Therefore, the reference values constructed from results produced by this work will be indicators of the costs supported by society due to the possible loss of these services.

Here, we will simply recall the foundations of the main approaches in order to specify their advantages and limitations for the practical measurement of biodiversity and ecosystem services affected by infrastructure projects. This will enable us to conclude this part on the potential of methods aiming to transfer the results obtained under certain conditions to new sites, on condition of a limited quantity of information.

These methods follow very different approaches, which are sometimes complementary, depending on the information used and the objectives pursued. We distinguish between the methods based on observable behaviours, assumed to reveal the agents' preferences and therefore the value they attach to the benefits of conservation, and the methods based on surveys by questionnaires that aim to collect the agents' statements about their preferences. Another dichotomy opposes the methods that can be used to directly obtain indications of value and those that involve deducing them indirectly from correlated information.

Table V-2: Diagram summarising evaluation methods for non-market goods

| | Revealed preferences | Stated preferences |
|---|---|---|
| Direct methods | Monetarisation at market prices<br>Costs of restoration, replacement<br>Costs avoided, productivity effects | Contingent evaluations |
| Indirect methods | Prevention expenditures and protection behaviours<br>Costs of moving<br>Hedonistic prices | Joint analysis<br>Contingent classification<br>Comparison by pairs |

*Source: CAS, Groupe biodiversité*

These methods aim to estimate the variations in surplus linked to changes in ecosystem services and biodiversity as accurately and robustly as possible. We should stress that the accuracy is generally not very great and that variations of plus or minus 20% between two measurements of the same variation in surplus are quite acceptable[1]. This report is not the place for discussing the theoretical foundations of environmental evaluation, which the reader can find in numerous manuals and works (notably, Desaigues and Point, 1993; Bonnieux and Desaigues, 1998; Perman *et al.*, 2003). We will try to limit the discussions as we first present the methods based on observed behaviours and then the approaches based on the statements of the people surveyed when faced with hypothetical scenarios.

[1] We should also note that a comparable level of error may also be made in the technical costs of an infrastructure project as well as its predicted level of use.

## 3.1. Methods based on costs

A first approach to the value of natural assets can be obtained from the estimation of the costs involved in maintaining the services provided by ecosystems. These may be replacement costs, restoration costs or relocation costs. The studies that have attempted to calculate the value of services provided by ecosystems on a large scale (Costanza *et al.*, 1997; Pimentel *et al.*, 1997) were largely based on this type of approach.

### *a. Monetarising the physical damage at market prices*

The simplest procedure in the face of real or potential damage to the quality of ecosystem services seems to be to describe the observed or predicted damage and attribute it a price. It was used in France to assess the damage related to the *Amoco Cadiz* accident (Bonnieux and Rainelli, 1990). However, the simplicity is only apparent for at least two reasons.

As it is a question of attributing prices to the damage measured in terms of lost use opportunities, this approach is mainly suited to evaluating direct uses. However, the prices of end goods and services do not necessarily give a pertinent indication of losses. If an oil slick makes the selling of a set of goods and services related to the sea and coast impossible for a certain period, it is *a priori* the net value of those products that is lost, i.e. the value that they would have added to the factors that their production would have consumed and which may not be lost[1]. However, we can imagine that in the case of an important change the factors used to produce those goods and services will not automatically find other uses and will, therefore, also be lost, at least in part. Therefore, the question of measuring losses of surplus must be resolved case-by-case, taking the specific circumstances into account.

It may be that the goods and services lost do not have a price. A kilogram of fish and a night in a hotel have a prices but a walk in a forest, even a very biodiverse one, is generally free, in the same way as bird watching or collecting mushrooms in state forests. It may even be very difficult to simulate a market, for example, for goods that cannot be appropriated. In practice, the evaluator can then attribute a price to the damage using scales, if there are any[2], or calculate the costs of restoration, if this avenue is open to him or her.

### *b. Evaluating the costs of restoration, replacement or relocation*

A pertinent basis for determining the cost incurred by the damage to an ecosystem may be by measuring what restoring it would cost. Ecological engineering has developed a set of methods that may be applied to a large range of inland and coastal ecosystems and we therefore have usable data available.

If the damage is irremediable, for example, because the biotope has been converted to a totally artificial use (roadway, industrial estate, etc.), we are in a situation where

---

[1] Counting simultaneously the price of the meals not served by the restaurant and that of the fish that have not been caught may lead to double accounting.
[2] There are some, for example, for the value of an individual of certain fauna species, published by the ONCFS.

the appropriate response is compensation. The notion of compensation has been defined and it should be understood that it cannot only be limited to acquiring a comparable ecosystem in another place. It is a question *a priori* of acquiring land whose ecological characteristics have been damaged (industrial wasteland, for example) and restoring those characteristics in order to compensate for the services destroyed by the re-creation of equivalent services elsewhere when that re-creation is possible[1].

The OECD manual (2002) stresses that the use of replacement costs is only *a priori* legitimate if three conditions are met:

- ecological engineering is really capable of producing systems providing the same functions in quantity and quality,
- ecological engineering is the least costly solution for providing these functions,
- its cost is less than the collective willingness-to-pay for maintaining these services.

In other words, the replacement costs only constitute acceptable measurements of the value of ecosystem services if the replacement solution is judged to be economically pertinent.

### *c. Evaluation of the costs avoided*

An alternative approach is to use the evaluation of the costs avoided by the existence of biodiverse ecosystem services, meaning, more or less directly, the cost of providing the services by artificial means if the asset was destroyed. The first example that springs to mind is that of the cost of water treatment stations that would be necessary to substitute for the purification of service provided by wetland areas in good working condition (this is the often quoted case of the protection provided by the Catskill Mountain for New York's water supplies). We could also mention the breeding of threatened species in captivity or the production of game to support recreational hunting.

### *d. Approach by the productivity effects*

This method considers ecosystem services to be inputs in a production function and estimates the consequences caused by a change in those services in terms of market products (agricultural production, forest products, etc.). Barbier (2000) used this method to calculate the value of mangroves from their contribution to the productivity of fisheries. The trickiest stage is to specify the distortion of the production function if the service, whose contribution we want to measure, is no longer provided in a similar way or at all. A simple form of this procedure was chosen by Gallai *et al.* (2009) for estimating the value of the pollination service at a worldwide scale. Chavas (2009)

[1] For example, the case of peat-bogs seems problematic. If the route of a motorway involves the destruction of a peat-bog, how could you create a peat-bog of similar maturity elsewhere? If it is impossible to really compensate for them should that lead to these ecosystems being made into sanctuaries?

used this procedure to measure the value of biodiversity, especially in the pollination service, on the basis of agricultural production functions.

The boundary with indirect methods becomes indistinct, because they fundamentally consist of evaluating ecosystem services on the basis of behaviours, therefore of expenditures agreed to on markets used as proxies for the non-existing markets for those services.

### *e. Advantages and limitations*

These approaches propose a set of indirect techniques that provide an estimate based on costs incurred for producing substitutes or for avoiding losses. Their pertinence is contingent on a set of factors and hypotheses:

- if the substitute goods are imperfect (for example, they only produce a part of the services provided by the ecosystem), the new valuation is partial and the result may be quite far from and *a priori* less than the real value,
- even if the substitute is complete, the fact that decision to replace is taken indicates simply that its cost is less than that of the service and not that it is the equal to it (but from an economic point of view, this restriction may be acceptable),
- finally, note that the agents likely to take the replacement decisions only consider *a priori* the benefits that they receive and it may well be that other benefits are lost that were enjoyed by people who do not have the technical or legal possibility of investing in their replacement[1].

## 3.2. Indirect methods based on the revealed preferences

The basic idea underlying the indirect methods is a monetary value for the changes in the level of services obtained from natural environments can be inferred from the behaviour of agents in markets related to them. For example, if an increase in the demand for fishing permits is observed following an improvement of the water quality in a river or a lake, we could try to use this variation to assign a value to the change in quality[1]. The evaluation of the damage, either real or expected, is based more or less directly on measuring the expenditure by the agents, in " substitution markets", such as prevention expenditure; hedonist prices deducted from the real estate market; costs of travel undertaken to obtain access to some services.

### *a. Protection expenditure approach*

We can try to deduce the value of biodiversity and ecosystem services from the expenditure made by the agents in an individual or collective capacity to prevent or attenuate the consequences of a loss of biodiversity or a reduction in the ecosystem services. This prevention or protection expenditure reflects the arbitrations that rational agents will agree to, in order to avoid harmful effects or to maintain the flow of a threatened service. However, this method is not used very much to evaluate

---

[1] Conversely, some authors note that if the replacement decision is taken by a public authority, it may be made for political reasons or under the influence of *lobbies*, with no guarantee that the benefits thus restored will be greater than the costs of the operations.

biodiversity. The efforts aimed at maintaining ecosystem services are primarily implemented by public agencies whose action may follow a more political than economic rationality and correspond rather to restoration or replacement costs.

### *b. Hedonist price methods*

This technique is based on the idea that the value of certain goods (real estate) reflects, beyond the particular characteristics of the good, the quality of its environment, both in regard to cultural and natural services. We can thus highlight the weight of the environmental quality in the pricing of goods that have been the subject of transactions, by means of a suitable econometric treatment. This weight constitutes a measure of the (capitalised) value or rather of the price of the environmental quality as reflected by the real estate market. This method is however criticised because it leads to a truncated measurement of the surplus variations related to the amenities taken into account.

This method has seldom been used to consider the benefits related to biodiversity or ecosystem services. A key question here is to characterise the ecosystem services or biodiversity that had an impact on the prices observed. Several studies relating to American cities have evaluated the value of wetlands or of open spaces in urban areas (Mahan *et al.*, 2000; Smith *et al.*, 2002) or the vicinity of riverain elements conserved in urban areas, in particular in desert areas (Bin and Polasky, 2005). In the metropolitan area of Tucson, Bark *et al.* (2009) highlighted the existence of differentiated purchaser preferences according to the quality of natural areas and certain characteristics of the bank vegetation and, in particular, the diversity of tree species, the presence of species that are typical for banks and moist environments, or the connectivity of bank habitats with those of higher grounds.

This technique seems well suited to measuring landscape amenities (Cavailhès and Joly, 2006; Cavailhès *et al.*, 2007). However, none of the 28 studies listed by Rambonilaza (2004) were based on hedonist prices. Other services, such as water or air quality were analyzed with this approach.

### *c. Travel cost method*

To use certain environmental assets (beaches, trout rivers, national parks, etc.), people may have to travel. As a first approximation, the expenditure can be assimilated as an access price that the users agree to pay to use the assets. To evaluate this expenditure, a direct survey is usually undertaken with the users, onsite. The travel costs integrate two main components: the transport expenditure and the value of the transport time.

The transport expenditure depends on the means of transport used, the number of passengers, the frequency of the visits, etc. The time of transport has been evaluated for a long time as a fraction of the hourly income of the user depending mainly on the selected character or constraints in the use of this time (between 1/3 and 1/4 of the wages). This approach was gradually given up and the evaluation is now done mainly from declared preferences or by modelling. The main difficulties encountered are still

---

[1] These approaches are based on the "weak complementarity principle" set forth by K.-G. Mäler (1974).

related to the value of time, to the treatment of multiple site situations, to "truncated" sampling, which cannot take into account the demand of non-visitors, to journeys having several objectives and to the choice of functional forms for econometric data processing.

This method is well suited to measuring recreation values and has been largely used in these cases. Smith and Kaoru (1990) had already listed nearly 200 studies, published or not, between 1970 and 1986, 77 provided them with, or were used to, calculate, a surplus variation and, thus, to highlight the influence of a set of variables such as the characteristics of the sites, the type of activity practicable and certain behaviour or decision parameters. The techniques have improved since, but the application of this method to biodiversity is limited to the evaluation of protected spaces or of charismatic species likely to cause travel behaviour. It has, in particular, been applied to the study of eco-tourism (Maille and Mendelsohn, 1993).

### *d. Advantages and limitations*

The methods based on revealed preferences do not take into account non-practical values (by definition) and cannot analyze attitudes in relation to risk in a relevant way. For these objectives, we would have to return to willingness-to-pays relating directly to the threatened assets. Since these preferences cannot be observed on any market, the only solution is to carry out investigations by questionnaires asking the subjects to declare what their behaviour would be if they were to face a hypothetical situation.

## 3.3. Methods based on declared preferences

The most mobilised methods, undoubtedly the most interesting ones because only they explicitly take into account values other than real use values, but undoubtedly also the most discussed and criticised, are based on "declared preferences"; that is to say, on willingness-to-pays or dichotomic preferences or other forms of expression of the interests gathered within the framework of standardised protocols and processing. A set of techniques are grouped under this name: contingent evaluations, contingent classifications, contingent notation, joint analyzis (*choice experiment*, *choice modelling*); classification by pairs and programme method.

### *a. Contingent evaluations*

Thought up by Ciriacy-Wantrup and implemented by R. Davis (1963) to estimate the value of forests as hunting grounds, the contingent evaluation method is based on investigations by questionnaire, carried out with a sample of agents, aimed at obtaining direct information on preferences, usually expressed in the form of a willingness-to-pay (WTP) to obtain or preserve a service. To do this, the subjects are asked to accept assumptions according to which they would have to make a choice: hypothetical scenario describing why the asset is threatened and, possibly, the nature of the actions to be undertaken. The questionnaire must specify the "payment vector", which specifies in which way the payment will be made and which must ensure that this operation is made more credible and persuade the subject directly concerned.

This approach has always been a cause of great mistrust, both for economists and other environment specialists, relating to its hypothetical character: the subjects who answer a questionnaire are not under the same conditions as they would be if they were making a true choice. It can of course be argued that this approach is not primarily different from carrying out market research for private goods, but, exactly, here it relates to services having public goods characteristics for which the preferences may not be expressed spontaneously as economic choices.

The literature regarding the contingent method has thus quickly been directed towards the location of multiple "biases" and ways of treating them:

- *Hypothetical bias*: related to the hypothetical situation, it is characterised *a priori* by an over-estimation of the declarations in relation to the revealed preferences, in particular for extreme values.
- *Strategic bias*: the people surveyed distort their answers to influence the results of the study and thus the resulting decisions, or their possible payment.
- *Informational bias*: related to the nature of the information transmitted to the people surveyed, likely to influence the preferences, but sometimes essential, in particular if the subjects have little familiarity with the assets.
- *Design bias*: related to the payment vector that the subject may believe he or she is able to avoid (Willinger, 1996) or a (badly) stated question (Bateman *et al.*, 2002).
- *Inclusion bias*: relates to the difficulty for the people surveyed to distinguish their WTP for the asset that the questionnaire relates to from broader overall categories. Variants: field bias (Kahneman and Knetsch, 1992), ranking bias, under-additivity bias, geographical bias (Dachary-Bernard, 2005, 2007).
- *Moral satisfaction bias* ("*warm glow*" effect): propensity of subjects to want to contribute to a "good cause" for the satisfaction that they derive from it, regardless of their real interest in the asset or the threatened service.

In practice, it is sometimes difficult to distinguish "biases" that are systematic errors dependent on the method and that involve a difference between the value obtained (by a contingent evaluation) and the "true value" for which we assume exists, and the "context effects", which translate the fact that the subjective value of the goods and services may depend on the situation or the circumstances[1].

The surveys often obtain a large number of null WTPs. These answers may have two types of significance: the subject is really indifferent to the change described or he/she considers that it is not up to him/her to pay ("zero protest"). Therefore, there is a vast literature on the treatment of failures to reply and/or false zeros which may lead to reconstructions aimed at correcting the average WTP.

Some work has confirmed the existence of a significant divergence between WTR and WTP which may go up to a factor of 10. The explanatory factors are related to the influence of the budgetary constraint and to the fact that the agents over-value the

[1] Brown and Slovic (1988) proposed a classification of the main context items likely to influence the value: the answering method (discrete choice, classification, notation or direct revelation of the WTP), order of magnitude, order effect (order of the questions), secondary information, consistency (for example, individual answers or as a manager), social context. G. Géniaux (1999) proposed a systematic analysis of this type of problem.

losses compared to the benefits, in particular because of the possibility of irreversible losses. This report led Hanemann (1991) to stress that this divergence showed *in fine* the fact that WTP and WTR do not measure the same thing.

An development of this approach is the programme method aimed at limiting the hypothetical bias by confronting the agents with more familiar choices than the declaration of a willingness-to-pay. The agents are faced with dichotomic choices between action plans affected by different costs/prices. *A posteriori*, the modellers can find a value for the natural assets concerned by suitable econometric data processing.

### *b. Joint analysis*

Joint analyses ("*choice modelling*", "*choice experiment*") are more recent approaches and *a priori* are more powerful because they confront the subjects with choices that are closer to the situations known by the agents. Initially conceived for marketing applications, the choice modelling method has aroused increasing interest in the field of the environment. Indeed, it is suitable for accounting for choices made between scenarios characterising various aspects of a project.

Any project can be divided into alternatives, described by combinations of attributes. The method is based on the Lancaster's theory (1966), according to which the utility obtained for a good or service is equal to the sum of the utilities obtained for its various attributes and characteristics. Using surveys, the people questioned are confronted with alternative descriptions of the project constructed by combining these various attributes. Descriptions of the alternatives are presented in a certain number of sets of choices including the *status quo* and at least one alternative option. The people questioned are invited to choose their preferred option from each set. Since one of the attributes is of monetary nature[1], the willingness-to-pay for the good and its various attributes can be inferred from the answers. Though the exercise is in general well understood and accepted, great efforts must be devoted to mastering the cognitive aspects in the project description in the choice exercise. This effort is all the more necessary when we are dealing with abstract concepts, such as biodiversity.

In order to improve the significance of the choice alternatives submitted to the subjects, these can be developed within the framework of a "*focus group"* made up for the occasion and to which suitable information is provided by "experts".

An alternative is the contingent classification method ("*contingent ranking"*). This approach is sometimes preferred because the authors consider that analyses based on an ordinal evaluation of the preferences will include less bias than cardinal measurements.

This method seems more tempting than that of contingent evaluation since it confronts the surveyed agents with choices closer to the situations that they would face in their real behaviour. However, a recognised specialist like N. Hanley does not consider that just modelling the choices should replace contingent evaluations but

[1] The exercise states for example that the cost of each change would represent an increase in local taxes for the residents.

rather that the two approaches will coexist according to the situations and the information available.

### c. Advantages and limitations

Only the methods based on stated preferences can provide indications of the values other than real uses, which partly explains their success, notwithstanding the difficulties encountered when trying to use them in a legal framework. As the real use values are usually not enough to economically justify conservation, obtaining an empirical measurement for wider values, reflecting effective interests, can be a determining factor.

This need does not imply obviously that the methods are reliable and robust. The contrary opinion prevailed for a long time (Hausman, 1993; Diamond and Hausman, 1994; Barde and Pearce, 1997; etc.) and still has serious arguments:

- the answers to questionnaires are generally over-estimated compared to the real behaviours, they can be unrealistic, not respecting budgetary constraints,
- the answers may contravene economic rationality and be influenced by effects such as the "*warm glow*"[1],
- the form of the investigation and, in particular, the way in which information is provided to the subjects may influence the results, introducing bias into the analyses and results likely to resist the techniques aimed at correcting them or at limiting them.

In 1992, the controversies raised by the *Exxon-Valdez* accident led the *National Oceanic and Atmospheric Administration* (NOAA) to gather a panel of prestigious scientists (Arrow *et al.*, 1993) who issued a series of recommendations for a reasonable use of these methods. Their main conclusions were the following:

- the samples must comply with careful rules,
- non responses must be minimized,
- the questionnaire must be tested (for example by a "*focus group*"),
- the investigation must be carried out personally (not by mail or telephone) and the questionnaire must be subjected to preliminary testing to limit the "investigator" effect,
- the results must not be reported without specifying the conditions under which they were obtained,
- the temporal variations in the results must be reduced, using the average of several independent studies carried out at different times[2],

---

[1] The *warm glow* effects in contingent evaluations can be identified from factor analyses and isolate a "cold" WTP which better corresponds to the economic concept of value (see Nunes and Schokkaert, 2003).
[2] The description of a temporal drift would be interpreted as a lack of robustness. A group gathered by *Resource for the Future* (Carson *et al.*, 1995) studied this issue for the case of the *Exxon-Valdez* without highlighting such a drift.

- the results of the evaluation must be sensitive to the importance ("*scope*") of the threats[1].

A recurring debate relates to the question of the pre-existence of preferences for carrying out their evaluation ("constructivism", Willinger, 1996). Faced with situations that are distant from the familiar choice situations, the suspicion that the preferences do not exist prior to the evaluation process raised the constructivism debate: Plott (1996) proposed a "*preference discovery hypothesis*" which must incite the people carrying out the evaluation to be very prudent in the development of the procedures for gathering information.

An example, mentioned by I. Bateman (2008), illustrates this concern well. It concerns a market study comparing two cosmetic creams that are similar in all aspects, except for their price and the fact that one is produced from palm oil obtained from "*tiger-friendly*" plantations. If this difference is only given in the shape of a text reassuring the buyer that measures are taken to protect the tiger population, the premium in terms of price difference can reach 15%. If the situation is dramatized by reporting that the tiger population was of 1,000 individuals in 1978 and only of 500 today, the premium increases to 21%. If it includes a photograph showing tigers in good health, it reaches 37%.

For methods, contingent evaluations or "*choice experiments*" that strongly depend on the form and the context, blunders or manipulation are always possible. However, the use of visual aids honestly reflecting the stakes and repeated measurements taken in contexts supporting the learning of the individuals questioned, contributes to the reliability and the robustness of the results.

The recommendations of the NOAA panel (National Oceanic and Atmospheric Administration) (Arrow *et al.*, 1993) created a slightly paradoxical situation: they legitimised the use of the contingent evaluation method, even for estimating non-use or passive use values, but specified a level of requirements making such studies particularly heavy and expensive to carry out, thus limiting their use to situations with weighty issues. Since methods based on revealed preferences cannot measure these values, the panel's recommendations thus reinforced the interest in meta-analyses for constructing transfer values, primarily from stated preferences. It was thus at the beginning of the 90s that the questions raised by this technique started to be the subject of an institutionalised debate and clarifications 2.

---

[1] The same group (Carson *et al.*, 1996) restated these recommendations and highlighted that correctly specified studies could pass such a test. V. K. Smith and L. Osborne (1996) carried out a meta-analysis on this issue underlining the difficulty in testing the rationality of the answers in an unambiguous way; but, taking as a starting point medical practices aimed at drawing generic lessons from a series of cases, they conclude with the feasibility of meta-analyses in so far as it is possible to bring studies to a common metric.

[2] In particular in the contributions to the special issue of the magazine Water Resource Research (within the framework of the *workshop* "*Benefits Transfer: Procedures, Problems, and Research Needs*", organised by the Association of Environmental and Resource Economists, Snowbird, UT, in June 1992.

## 3.4. The question of value transfers

The implementation of most of the evaluation methods, in particular from stated preferences, is very expensive and requires time. Also, from the point of view of aiding with concrete decisions, it appeared necessary to find means of “standardising” these values in order to be able to use them in other situations. The transfer of benefits or environmental values is thus a technique by which the results of monetary evaluations of environmental assets are applied in a context different from that or those in which they were made. This technique is controversial, in particular owing to the fact that a certain number of scientists and decision makers have reservations about the relevance and validity of the transferred values to justify the importance of the environmental issues in the socio-economic evaluation of projects (Brouwer, 2000).

### *a. Value transfer: principle and validity*

Two approaches are possible: either the unit value obtained on a “*study site*” is transferred to a site concerned with the issues of a decision (“*policy site*”) with similar characteristics; or a demand equation is transferred by applying the coefficients estimated for one site to the value taken by the corresponding variables on the other site.

To test the quality of a transfer technique, the non-commercial values are considered on both sites (study site and policy site) using the primary data gathered from both sites; then, the results obtained with the site's data are compared with the result which would be derived from the use of the data on the other site. If the results are not statistically different, it is assumed that the transfer of benefit is valid. If not, the analyst must examine the sign of the biases and their size to try to attenuate them.

Loomis (1992) tested the performance of the transfer of a demand equation for recreational fishing in three American States resulting from the evaluation by travel costs carried out in an identical way. The transfer of the demand function led to differences of 5% to 15%, while the transfer of the average travel values led to much larger variations. Other studies showed the importance of gathering data according to similar procedures (including at the same date, in particular for the recreational demand), for sites with the greatest similarity.

Kirchhoff *et al.* (1996) empirically validated the fact that the transfer of values in the form of a function (of a set of explanatory variables) was more robust than the transfer of average unit values for a certain type of sites. This was not unexpected, but the difficulties of the exercise, in particular, in terms of standardisation, justify raising the question. They also underline the fact that the circumstances in which the transfers are valid and relevant for the decisions could be quite limited and that the errors resulting from the transfers may be considerable, even between sites presenting quite similar amenities.

One of the recurring problems encountered in the transfer of values from meta-analyses is the strong variance of the results which seems to be largely related to the “design” of the surveys (rather than to the characteristics of the sites or their uses).

Brouwer (2000) consider that the value transfer tests have not justified their practice and that many transfers published in economic reviews do not fulfil the minimal robustness criteria. Highlighting the weight of certain factors, he suggests restrictive orientations on the acceptable practices which in particular distinguish the transfers made based on one only study, from those based on multiple studies. These indications are summarised by a seven point protocol:

1. clearly define the environmental goods and services,
2. identify the stakeholders,
3. identify the values for the stakeholders,
4. involve the stakeholders in the validation of the monetary evaluations,
5. pay great attention to the selection of the studies,
6. take into account the effects of the evaluation method,
7. involve the stakeholders in the aggregation of the values.

Obviously, this protocol cannot be systematically followed, but the idea of associating the "*stakeholders*" at various times in the exercise is also a means of allowing them to express their perception and, if required, to build a shared representation, which is one of the motivations of a deliberative approach of the evaluation.

In 2006, the *Ecological Economics* magazine opened its pages to a group of recognised specialists in evaluation and transfer techniques to allow them to confront their points of view on the strengths, weaknesses and possible improvements of these approaches. In a general introduction, Wilson and Hoehn (2006) specified what currently seems still to be the main issues raised by the validation and standardisation of these approaches and, in particular, the importance of an interdisciplinary collaboration, both for social sciences and for natural sciences. Loomis and Rosenberger (2006) stressed the importance of anticipating the future use for transfers in the design of evaluation studies. Considering the importance of the design for limiting the biases related to the evaluation techniques, it is important that these biases are taken into account in a way that facilitates the transfer of the results to different sites. In the same sense, Bergstrom and Taylor (2006) considered that meta-analyses can only become useful tools for the transfers if they are founded on studies carried out following systematic protocols for the development of models, data gathering and analysis, as well as their interpretation. This still involves multiplying the validation tests. The strength of the meta-analyses lies in their capacity to combine and to summarise significant amounts of information, but this may be their weakness if it involves the loss of significant elements with space or temporal specificities in the aggregation process. For the transfer of values related to ecosystem services, Hoehn (2006) stressed the importance of the selection effects and suggested a two stage procedure, adaptable to the incomplete panel data, to locate and test the importance of sampling biases, which divided the results of a regression by four, avoiding this bias in the choice of the sites studied[1].

---

[1] In this same number and from a critical point of view, Spash and Vatn (2006) underline the existence of difficulties which must be interpreted within the more general framework of informational questions concerning environmental problems and suggest alternative approaches, more capable of dealing with the multiple environmental values stressing multicriteria approaches and the importance of participative and deliberative institutions.

In 2006, the *Environmental Protection Agency* (USEPA) gathered a work group which evaluated and summarised 140 meta-analyses resulting from 125 studies, some published and some not, in 17 fields of the resource and environmental economics. A critical summary of the work of this group was recently published (Nelson and Kennedy, 2009) presenting several generic models of meta-analyses and identifying five points on which econometrics must particularly take care to respect good practices: the selection of the samples, the source data, the heterogeneity of the primary data, the heteroscedasticity problems and the independence of the observations of primary studies. The article concludes with a "good practices" guide and a discussion of the main methods used to transfer the environmental values resulting from meta-analyses.

Nowadays, economic literature offers a great number of meta-analyses for a number of goods and services related to biodiversity and ecosystems:

- value of protecting mangroves: Barbier (2000),
- forests: Bateman and Jones (2003), Willis *et al.* (2006), Brahic and Terreaux (2009), Zandersen and Tol (2009),
- wetlands: Brouwer *et al.* (1999), Woodward and Wui (2001), Brander *et al.* (2006), Enjolras and Boisson (2007),
- landscapes: Rambonilaza (2004),
- threatened species: Loomis and White (1996), Richardson and Loomis (2009),
- nature conservation: Tuan and Lindhjem (2008),
- value of biodiversity: Nijkamp, Vindigni and Nunes (2008).

The multiplication of this work has been largely facilitated by the development of international databases that compile the work published, but also the evaluations which are not and would otherwise remain difficult to identify and to obtain.

### *b. The development of databases*

After at least four decades of research and studies on the valuation of natural assets, the advantages of compiling databases has become increasingly obvious and has led to the creation of databases of which the most well known is undoubtedly the *Environmental Valuation Reference Inventory* (EVRI). The EVRI is defined as "a repository for empirical studies relating to the economic value of environmental benefits and effects on health". It was conceived as an instrument aimed at helping people who wish to evaluate policies using the benefits transfer method. By carrying out a benefits transfer using the EVRI, you can avoid setting up a complete valuation study".

The summaries of the EVRI valuation studies provide the broad outlines of the relevant questions about the evaluations and results that a researcher must obtain in order to identify the best studies applicable to a possible benefits transfer. There are six main information categories included in more than thirty fields.

1. Subjects of study - basic bibliographical information.

2. Area of the study and characteristics of the population – information on the place of the study and the data relative to the place and the population.
3. Environmental key points of the study – the sectors where the environmental assets evaluated are described, factors of stress on the environment and the specific objective of the study.
4. Study methods – the technical information on the study in question, the specific techniques used for the results obtained.
5. Estimated values – the monetary values represented in the study and the specific unit of measure.
6. Summary in other languages – a summary of the study is available in English and French.

Currently, the EVRI has more than 2,000 studies, classified according to several category systems. Biodiversity does not appear as such, but ecological functions, species and habitats can be found. The database can thus be used to prepare meta-analyses and some analyses refer explicitly to it (Rambonilaza, 2004; Zandersen and Tol, 2009). Developed by *Environment Canada*, the EVRI was the subject of an agreement between the United States, the United Kingdom, Australia and France. Negotiations are underway with New Zealand and Switzerland.

The EVRI (www.evri.ca) is accessible free of charge for the citizens of the Member States of the "EVRI Club". There are other databases, more or less freely accessible: the Australian base *Envalue* (www.epa.nsw.gov.au/envalue), the *Ecosystem Services Database* (www.esd.uvm.edu) developed by the Gund Institute for Ecological Economics of University of Vermont or the *Review of Externality Data* of the European Commission (www.red-externalities.net). There are several other initiatives (McComb *et al.*, 2006), among others the *Biodiversity Economics* database of the UICN – The World Conservation Union – WWF (www.biodiversityeconomics.org) and *Ecosystem Valuation* (www.ecosystemvaluation.org) developed by D. King (university of Maryland) and M. Mazzotta (University of Rhode Island) with the support of the USDA and the NOAA. We should also mention two databases developed in France, one concerning water (www.economie.eaufrance.fr/), the other concerning forests (http://lef.nancy-engref.inra.fr/).

## 3.5. To conclude

Biodiversity and the ecosystem services are a "complex asset" (the agents obtain utility from them in several ways) with which the agents are not necessarily familiar. Hence, there are reservations about basing the measurement of the social value of these assets only on the preferences of the agents. The idea is thus to associate information from the agents-citizens with other information resulting from the skills of "experts". The problem is that there is no general method *a priori* to incorporate this information within a consistent framework with unambiguous significance. This can however be done by an approach combining a development phase, in which detailed information is provided to a group of agents in the presence of experts, thus enabling them to define the issues together, with the subsequent carrying out a survey by questionnaire, dichotomic *a priori*, leading the agents to declare preferences within a framework that has become more familiar.

We should stress here that the fact of evaluating biodiversity (or global ecosystem services or on a country or continent scale) is a very different question from that of seeking reference values to evaluate the impact of a localised project. The first question raises fundamental problems, in particular related to the trajectory of human societies and to the unforeseeable character of long-term changes caused by large scale transformations of the biosphere (see for example Vitouzek *et al.*, 1997 or Diamond, 2000, 2006). The exercises that several groups of scientists have undertaken (Costanza *et al.,* 1997; Pimentel *et al.*, 1997) are based on very simplified assumptions[1] and only aim for conservative values[2]. More recently, more institutional initiatives (Braat and ten Brink, 2008; Sukhdev, 2008) have endeavoured to define more comprehensive approaches but only obtain numerical results at the cost of drastic simplifications.

Our purpose is more limited, it is a question of defining indicators having a price dimension, reflecting the losses of services undergone by society due to the destruction, degradation or disturbance of ecosystems by defined projects. As limited as it may be, this question nonetheless raises formidable difficulties:

- how can the impact of a project on evolving ecosystems be identified, including those under the influence of other human actions?
- which values should be taken into account?

---

[1] The notoriety of the study by Costanza *et al.* (1997) for non-economists obviously did not protect it from virulent criticism by many economists (Toman, 1998; Bockstael *et al.*, 2000, etc.). By showing that a conservative estimate of the services contributed by the ecosystems represented a value of an order of magnitude of 1 to 3 times the gross world product, it however constituted an important moment in the description of the issues.

[2] The *a priori* illegitimate character of an extension of local results to large scales is the first criticism that can be directed at the studies, which calculated values for all of the services rendered. However, these two studies (Costanza *et al.*, 1997; Pimentel *et al.,* 1997) are based mainly on simple approaches in terms of cost (replacement cost, impact on the production functions) and thus obtain a value of contribution to the wealth produced and not an evaluation of what the losses of these services would represent. An completely different approach to the evaluation of the ecosystem services on a global scale was proposed by Alexander *et al.* (1998), who considered that this value is raised by the world gross product from which the maintenance expenses of the world population are obtained. Perhaps this approach highlights the fact that, vis-a-vis of such important issues, the analysts no longer know to which situation they should refer.

- how is the population concerned (and aggregation issue) identified?
- to which surface area units should they be applied?
- how should the responsibility for costs caused jointly by several human actions be shared?

The strengthening of the agents' familiarity with both the assets to be evaluated and the WTP determination methods is a need argued by many authors. Schläpfer (2008) suggests three solutions:

- to help the agents to formulate consistent preferences revealing hypothetical costs for realistic political scenarios in a credible order of magnitude for the subjects (for example, by relating the prices mentioned in the dichotomic choices to amounts already paid, such as a percentage of increase in the taxes),
- to offer the subjects the possibility of communicating with peers and experts,
- to give access to a credible help generated by political competition with which the subjects have a certain familiarity and to present the options by locating them in relation to the positions adopted by certain well identified interest groups.

The question the heterogeneity of the agents' preferences, depending in particular on their income (Jacobsen and Hanley, 2009) and their uses, must sometimes be the subject of specific treatment. This heterogeneity can concern contradictory preferences for elements to which some agents attribute positive values, whereas others perceive them mainly as the cause of problems. With regard to this subject, everyone can think of conflict situations related to the return or reintroduction of some large predators (wolf, lynx and bear). But the wetlands are also associated with harmful effects, such as the presence of mosquitoes[1] (Westerberg and Lifran, 2008), and a larger biodiversity can also mean the multiplication of self-propagating weeds and other crop pests (Zhang *et al.*, 2007). For agents whose interests are particularly affected (farmers, tourism economy, etc.), an increase in biodiversity can thus represent, at least in the short term, an increase in costs or losses.

In addition, we should stress that the change of assignment of a unit of area can have direct and indirect effects on biodiversity with cascade effect mechanisms (see for example Kinzig *et al.,* 2005) or of "nonconvexities" (Dasgupta and Mäler, 2003) that are difficult to summarise in an on site evaluation, except by providing the subjects with suitable information about all of the foreseeable consequences.

## 4. Summary of the main results published

In his article published in *Encyclopedia of Biodiversity*, Heal (2000) proposed that we should consider biodiversity to be consumer goods (a "*commodity*"), which leads him

[1] The presence of mosquitos is a typical example of ambivalent effects, because they also constitute the food of fish sought by fishermen. And diversified agricultural landscapes shelter mosquito population controlling species.

to successively examine how biodiversity supports the productivity of the economy, constitutes a form of insurance, increases the quantity of genetic information, supports ecosystem services and is related to commercial products. The minimal report is that biodiversity has a value for multiple reasons that we will explore using a dual reasoning: value of which aspects or components of biodiversity and value for which types of uses.

In this study, we are interested both in the value of biodiversity and that of ecosystem services, knowing that these two dimensions of our societies' dependence on Nature are closely related. We will successively review the analyses of the diversity of living beings, genetic resources, species, ecosystems and habitats, ecological functions and, finally, services with specific development for landscape amenities.

## 4.1. Evaluating the diversity of biodiversity?

These are primarily theoretical works that tried to deal with the question of the value of diversity, which can be explained mainly by the difficulty in working out a procedure for collecting stated preferences for assets that the subjects have necessarily much difficulty in grasping (even the specialists do not always share a consistent representation of these questions).

Undoubtedly the most well known are the articles of M. Weitzman (1992, 1993, 1998) whose approach is based on the possibility of measuring the genetic dissimilitude between two species[1] (but the replacement of the concept of species by populations or other entities, including individuals, would not change the reasoning) by ADN-ADN hybridization. Diversity is then measured by an iterative dissimilitude calculation procedure (between the added species and the closest among the species already present). This gives a function $D$ that Weitzman (1992) interprets as a diversity value, without clarifying for what and why diversity would be useful or desirable; diversity is considered important in itself. The value of a species or a set of species is then measured, within the framework of what Weitzman (1998) qualifies as the “Noah's Ark problem”, i.e. “how to preserve biodiversity as well as possible on a restricted budget”, by the sum of two magnitudes: $D + U$, where $U$ represents the utility derived from the existence of the species or entity in question, for the citizens.

The idea of “correcting” the utility obtained by the citizens from the existence of elements of the biodiversity by associating an expert component to it correctly conveys the idea of merit goods that we presented above. The key question, which the economic literature apparently cannot answer, concerns the method that could be used to add these two heterogeneous magnitudes[2].

This approach was widely criticised as being inapplicable to concrete problems both because of the information requirements and the enormous amount of calculations necessary, since the number of species is rather large (without reconsidering the limits

[1] The idea of basing the evaluation on cardinal dissimilitudes is unreasonably demanding in information and in computing power. Bervoets and Gravel (2007) proposed an approach based on ordinal dissimilitudes, assuming that the decision makers are able rank the differences, but without requiring measurement.
[2] Weitzman (1993, 1998) does not discuss this issue and it is hard to imagine, *a priori*, that the “experts” attach a monetary measure to the dissimilitude.

of the concept of species). Weikard (2002) however showed that the approach had a certain practical relevance if it was no longer applied to either species or populations, but rather to ecosystems.

Here, one can mention the work of Nehring and Puppe (2002), who constructed an economic value for diversity from the sum of the attributes, assuming that the decision maker has identified a series of attributes to which he or she attaches importance. This is subjective and the decision maker must indicate the importance attached to each attribute by means of a numerical value. The diversity of a sample is then measured by the sum of the values of the attributes. This approach has apparently never been concretely applied.

Empirical measurements of the value of the diversity of biodiversity are very few because the agents (except for enlightened naturalists) generally do not have sufficient understanding of what biodiversity is and why it could be important for them: their behaviours thus do not reflect these values and contingent evaluations cannot find them.

In a review of a set of analyses relating to genes, species, habitats and functions, Nunes and Van der Bergh (2001) highlighted the difficulties of these exercises and the heterogeneous character of the analyses, the pre-eminence of the methods based on stated preferences, which are the only ones appropriate for taking into account the non-use values and the incomplete character of the services considered, leading them to conclude that the empirical evaluations constituted "at best" the lower limits of the full value lost during changes in the biodiversity.

Christie *et al.* (2006) tried to evaluate "the diversity of biodiversity" using the *choice experiment* method. They used *Focus groups* to identify the concepts of biodiversity to which subjects allotted importance and to describe them in understandable terms that made sense to the subjects. The modelling of the choices was used to highlight the values that the subjects allot to the various attributes of diversity (familiarity with the species, scarcity of the species, habitat, functioning of the ecosystems). A contingent evaluation considered the willingness-to-pays in parallel for conservation policies such as agri-environmental measures or habitat re-creation operations. The results highlighted a positive value for several attributes of diversity (not all); but little interest for the means implemented to preserve it. The authors conclude from this that the definition of conservation policies should perhaps combine an economic analysis of the amounts to be allocated to the conservation of the biodiversity, and a choice of the means to be used for the greatest practical effectiveness on an ecological basis. An additional conclusion was that the workshops organised for their work had allowed exchanges and an enrichment of information that significantly contributed to reduce the variability of the estimates.

## 4.2. Evaluating genes: the case of bio prospection

The argument that biodiversity could be preserved, at least to some extent, by the commercial values of which it is the support, has frequently been put forward, in particular at the beginning of the 90s[1]. The biodiversity of ecosystems increases the chances of finding potentially useful things, in particular for the pharmaceutical industry.

One of the most mediatised examples of biodiversity valuation is the agreement made in 1991 between the pharmaceutical company Merck and the National Institute for Biodiversity of Costa-Rica (INBio): by paying 1.1 million dollars, Merck obtained for two years (the contract was then renewed in 1994 and 1996) exclusive rights to explore and exploit (giving rise to royalties) the pharmacological properties of the plants and micro-organisms in the country's 105 National parks[2].

On this question of "bio prospection", many studies have tried to consider the medicinal value of a plant that has not yet been tested, which can be regarded as a potential value of an element of remarkable vegetable biodiversity. First estimates of the value of preserving a species for a pharmaceutical use consisted in multiplying the probability of finding a substance of commercial interest by the net average income derived from the products having succeeded; these led to very varied values (from a few tens of dollars to several million; see Perrings, 1995a; Polasky *et al.,* 2005).

Simpson, Sedjo and Reid (1996) criticised this approach, which estimated an average value and not a marginal value. However, since multiple species may contain the same active ingredient, it is probable that the marginal values may have been much smaller than the average values. They thus developed a sequential research model taking into account this redundancy between the species. In their model, all species have an equal probability of containing an active ingredient and are tested with a cost *c*3. The value of the final species is the expected profit of the test, supposing that all the other species have been previously tested without success. With realistic values, their calculation applied to plants with flowers leads to a marginal value of 9,430. From a simple relation between the number of species and surface, it is thus possible to calculate the value of one hectare, which varies between 0.20 dollars in California and 20.63 dollars for the west of Ecuador.

Rausser and Small (2000) proposed a better ordered prospection model, considering that the prospectors had information allowing them to not allot the same chances of success to all of the species, which increases the value of the samples appreciably. The value of the most promising hectares of the west of Ecuador thus amounts to 9,177 dollars. Costello and Ward (2003) however highlighted that the differences between the results obtained by Simpson *et al.* (1996) and Rausser and Small (2000) were not mainly related to the effectiveness gained in the organisation of the bio-prospection but rather to the different assumptions regarding other parameters conditioning the value for the pharmaceutical industry 4.

[1] In particular within the framework of the debates about the Convention on biological diversity whose main issue was finally the sharing of the benefits of genetic resources. The idea that bio-prospection involves important issues leads to granting a great consideration to the access terms (Trommetter, 2005).
[2] Which, according to our calculations, allots a value to biodiversity of less than one dollar per hectare and per year.
[3] Polasky and Solow (1995) had used a similar model to evaluate a collection of species. They had shown that the existence of imperfect substitutes (the active ingredient is not exactly the same one) the value of the marginal species did not decrease so quickly.
[4] Re-doing the calculations with the same parameters, they found for one hectare of very bio-diverse forest in the west of Ecuador, a value of 9,177 dollars per hectare if the prospection is ordered and 8,840 dollars with a random research.

The summaries presented in the reports of the OCDE (2001) and by Pearce and Pearce (2001) show that the value assigned to vegetable bio prospection varies according to the authors between 200 dollars and more than two million dollars per plant. The reasons for this variation are due mainly to two factors:

- The probability of finding an interesting plant among all the plants tested varies according to the authors between a chance in 10,000 to more than one chance in 100,
- the value assigned to a plant leading to a medication, namely the annual profit related to this medication, is estimated at between 250,000 and 37.5 billion dollars. One of the leading causes of the variation is related to whether we consider the medication is only from the point of view of the profits for the company or we take broader elements into account, in particular by valuing the number of lives saved.

This means that the value of promising zones in terms of biodiversity will be considered in a range going from less than 1 dollar (which was the case of Merck for a contract that can be interpreted as having bearing only on the option values) to close to 10,000 dollars per hectare.

The recent literature review of Sarr *et al.* (2008) replaces the question of the value of the genetic resources for R & D within the framework of the debates on sustainability: up to what point is society able to invest now in conservation to avoid the occurrence of future biological problems? They show that the Weitzman (1992, 1993) classification indices reflect current society's issues and are not very relevant for anticipating uncertainties in the future. The sequential approach of Simpson *et al.* (1996) seems to reduce the future risks, on condition that they are well identified and controllable. The relevance of the bio-technological approach of Goeschl and Swanson (2002) is also limited to problems with no real uncertainty and to the maintenance of resources that can control these problems. Uncertainty is at the heart of the problem analysed by Kassar and Lasserre (2004) which leads to allotting an additional value to the options that do not limit future choices. Their conclusion (Sarr et al., 2008) is that the models tend to dismiss the pessimistic point of view about the ability of future technical changes to solve sustainability issues. The value of the genetic resources thus appears to depend *in fine* on our beliefs about the capacity of the present's objectives to anticipate the risks and uncertainties of the future.

## 4.3. Evaluating species

There is a large collection of studies on the value of species, in particular related to the existence in the United States of the *Endangered Species Act* (law on threatened species, of 1973), which led to this type of work being done from the point of view of budgetary rationalisation (see for example Brown and Shogren, 1998; Metrick and Weitzman, 1998). In addition, the contingent evaluation method is well suited to the evaluation of willingness-to-pays (WTP) for the conservation of endangered species, especially if the subjects with whom the surveys are carried out can visualise them easily. A lot can thus be found in the scientific magazines, with a difficulty that could be described as “publication bias”. Each study must be presented in a limited format that leads the authors and the reporters to be more interested in the methodological questions 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 second part of the question, which would be to consider the population concerned in order to be able to calculate a social WTP, which it would then have to be related to the objective (to preserve the species in the world, the area, or a given specific surface), is generally absent.

Most of the work concerns charismatic or emblematic species. It should be stressed that this concept does not only depend on ecological aspects but also integrates a sociocultural 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 V-3 a certain number of studies relating to threatened vertebrate populations, for which the willingness-to-pay per household and per year[1] was estimated for various protection measures of these species. 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 we only give the average values, in current currency).

**Table V-3: Willingness to pay for various emblematic vertebrate species (in dollars per site and per year)**

| Group | Species | Place of the investigation | WTP ($) | Reference |
|---|---|---|---|---|
| Mammals | Rhinoceros | The U.K. for Namibia | 54-65 | OCDE, 2001 |
| | Wolf | Sweden | 126 | Loomis and White, 1996 |
| | Grizzly bear | The United States | 46 | *Id.* |
| | Sea otter | The United States | 29 | *Id.* |
| | Gray whale | The United States | 26 | *Id.* |
| | Urial | The United States | 21 | *Id.* |
| | Caribou | Canada | 14-98 | Anielski and Wilson, 2005 |
| Birds | Northern spotted owl (*Strix Occidentalis*) | The United States | 70 | Loomis and White, 1996 |
| | Whooping crane (*Grus Americana*) | The United States | 35 | *Id.* |
| | Red-Cockaded Woodpecker *(Picoides borealis)* | The United States | 13 | *Id.* |
| | Bald-eagle *(Haliaeetus leucocephalus)* | The United States | 24 | *Id.* |
| Reptiles | Sea turtle | The United States | 13 | Loomis and White, 1996 |
| Fish | Pacific salmon | The United States | 63 | Loomis and White, 1996 |
| | Cutthroat trout (*Oncorhynchus Clarkii Utah*) | The United States | 13 | *Id.* |
| | Atlantic salmon | The United States | 8 | *Id.* |
| | Squawfish | The United States | 8 | *Id.* |
| | Stripped shiner | The United States | 6 | *Id.* |

*Source: CAS, biodiversity group*

[1] These are random samples of households, which do not distinguish between those that possibly use these resources (practical values) and non-users, who will express non-use values.

The values are relatively close, with extremes fluctuating in a ratio of 1 to 15 and some factors explaining these variations: thus, a hierarchy privileging mammals and birds compared to fish can be observed, with large amounts for very emblematic species like Pacific salmon among the latter, which contrasts with the low values for closely related species similarly threatened, but sedentary, in the same zone. Also note, on a purely anecdotal basis, the fact that residents of the Atlantic coast (the study was made in Massachusetts) seem to grant much less value to "their" salmon than those of the Pacific coast. Loomis and White propose 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 (the proximity paradox appears again) and 42 dollars if a single payment is proposed (which in fact corresponds to a reduction in the total payment).

Brown and Shogren (1998) showed that the results of Loomis and White (1996) result 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.

An extension of this approach was however proposed by Allen and Loomis (2006) who presented a WTP estimate model derived for "ordinary species" when these are the prey of protected species: knowing the feeding habits of the predator and the energy value of each prey, the implicit value granted to each prey can be deduced. For example, from a WTP estimate of 18 dollars per site to preserve a population of 12 golden eagles in Idaho (that is to say a willingness to pay 8.06 million dollars by all of the inhabitants of this State), the authors estimate that a California hare has an implicit value of 562 dollars per piece, that a marmot has a value of 861 dollars and that a pheasant has a value of 381 dollars!

Kotchen and Reiling (2000) explored the influence of the attitudes towards the environment on the answers to a contingent evaluation related to the protection of two species, the peregrine falcon *(falco peregrinus)* and the shortnose sturgeon (*acipenser brevirostrum*). They showed that the "pro-environmentalists"[1] are significantly more favourable to protection activities and that their average willingness to pay is significantly higher, in particular for non-use values related to ethical reasons.

Chambers and Whitehead (2003) studied the willingness-to-pay for a wolf management plan and the management of the damage related to this animal in Minnesota by the contingent method. Null answers are distinguished econometrically from *"I don't know"* answers, which constitute a large fraction of the answers. They show that both programmes generate benefits higher than their costs.

Of the 37 studies listed and standardised by Brahic and Terreaux (2008), the willingness-to-pay for the conservation of various species goes from 5 to 200 $\text{Euros}_{(2008)}$ per household and per year, with a modal zone between 12 and 40 Euros. These figures result almost exclusively from contingent evaluations and are thus

[1] The attitudes are measured according to a standardised scale (called the *New Ecological Paradigm*).

affected by biases related to this technique, in particular the inclusion or scale bias that has already been pointed out by the critics of Loomis and White[1].

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 VET 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. The model is used to test a transfer of value and the rate error for the original studies varies between 34% and 45%.

These methodological refinements do not completely eliminate the problems involved in the use of these values as reference values, in particular because of the presence, in the variables controlling the transfers, of variables related to the design or the conditions of carrying out the study, beside variables related to the subject evaluated. In addition to the additivity problems previously mentioned, the use of these values often brings the problem of disproportion, even of disconnection between the location and the sometimes limited character of the territory to be protected[2], 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.

Indeed, whereas for elements of ordinary biodiversity, one can consider that it is mainly the users living in or visiting the site who are both the ones who benefit from these services, through use values, and those who are likely to finance their management costs, these two populations are largely different – and even sometimes antagonistic as we have noted – when it is a case of remarkable biodiversity like the charismatic species 3 , which introduces undeniable political and ethical dimensions into the debate.

Moreover, to take an order of magnitude, it would not be outrageous to suggest that the population likely to pay ten dollars per year for a endangered large mammal species is of the order of a hundred million people on the entire planet. If the area to be protected is around a few hundreds of $km^2$, this leads to a WTP per hectare and per year higher than 10,000 dollars, which, for only one species, would exceed all the estimates based on ecosystem services and would even make it possible to oppose any human development activities that may affect this area.

We may be delighted by this argument which may seem decisive and would allow the areas concerned to be "sanctuarized", but it has its Achilles' heel: if we ask the same

---

[1] Desvouges *et al.* (1993) also obtained similar willingnesses-to-pay to save 2,000, 20,000 or 200,000 birds (80, 78 and 88 dollars respectively).

[2] This is not the case for large animal species which, even in reduced number, still have important requirements in terms of habitats. However, the problem of "disconnection" is also present.

[3] To integrate robust elements of economic value in these situations, we could, *at least*, develop approches specifically identifying use values, for example through the local economic consequences related to various labels emphasising elements of remarkable biodiversity (National parks, Great sites of France, UNESCO World Heritage, etc.).

citizens if they consider it justifiable to spend a billion dollars per year of public money to protect the same species, we can expect, at the very least, tepid answers! It would thus be better to replace these studies of only one emblematic species by more global solutions, presenting various conservation programmes for a set of species to be protected in a given environment.

The case of invasive or invading species brings up multiple problems, in particular when these species give rise to ambivalent perceptions by the populations. This has been the subject of evaluation work (for example Thomas *et al.*, 2006), but especially of analysis of the issues and means to manage these problems. The latter are generally not related to infrastructure projects and this will thus not be considered here. Interested readers can find a quite complete summary in the contributions to the work edited by Perrings *et al.* (2000), in the special issue devoted to this subject by the *American Journal of Agricultural Economics* in 2002, or Polasky *et al.* (2005).

## 4.4. Evaluating ecosystems or habitats

Environments are potentially multi-use and this plasticity seems to increase with their "naturalness". There is a set of studies that present the value of certain types of ecosystems as the sum of the values of the services that human societies obtain from them. The most interesting cases are those where the same study breaks up a total value, as is proposed in the last chapters of the summary work on the value of Mediterranean forests (in 18 countries) coordinated by Merlo and Croitoru (2005), within the framework of the European project MEDFOREX (*Mediterranean Forest Externalities*). The multi-functionality of the Mediterranean forest, considered to be a biodiversity "*hotspot*", is supported by full value measurements that on average lead to allotting 35% of the total to the production of wood, 21% to hunting, 16% to other recreational activities and 38% to option and existence values. This work highlights a strong overlap between the various material and non-material services produced, products with associated management systems that have variable objectives and which are unequally efficient. The French case, studied by Montagné et al. (2005), offers a division which seems very different, although the categories are only imperfectly outlined: wood represents less than 10%, collection products, 3%, hunting 1%, but other recreational activities reach 50%, the various protection services would amount to 15% , 10% are allotted to the storage of carbon and the demand for biological diversity is estimated at 10%, of a total which amounts on average to 240 Euros per hectare.

But even when almost all uses are prohibited, the reasons for conservation can be multiple. In a study related to an integral reserve ("*wilderness area*") in Colorado, Walsh *et al.* (1984) separately analysed the willingnesses to pay for the option value, existence value and legacy value, which contribute in a balanced way to a full preservation value, estimated between 14 and 32 current dollars.

A Finnish study (Li *et al.*, 2004) relating to the willingness to receive of 4,000 people in compensation for a reduction of the conserved areas, resulted in an average WTR of almost 700 Euros, whereas the average WTP for an equivalent increase is of 160 Euros (and the average WTP was null).

A cost/effectiveness reasoning was applied to the definition of efficient policies for habitat conservation. The question is thus to select the relevant sites, which most of the studies brought back to a problem of “maximum species coverage” within a network of sites[1] (Church *et al.*, 1996). Ando *et al.* (1998) integrated the value of the land into this problem as an indicator of the conservation's opportunity costs. The problem can be adapted to integrate approaches for the diversity other than the species. Polasky *et al.* (2001) showed that the maximisation objective of phylogenetic diversity led to a scheme very closely related to that of species maximisation… because of the strong correlation of the objectives.

Tuan and Lindhjem (2008) carried out a meta-analysis on a hundred studies that evaluated nature conservation in Asia and Oceania. They divided their database into two groups: studies relating to the protection of endangered species and those relating to more general conservation programmes. They showed that the studies relating to the species are more homogeneous and comply with theoretical and empirical expectations: the WTP are higher for mammals than for other species. They used their model to predict the results of studies not appearing in their database and obtained a smaller median/average error rate for the species (24/46) than for the conservation programmes (46/89). Their conclusion is that it is preferable to control heterogeneity in the regressions and the sensitivity analyses rather than to *a priori* exclude studies on criteria difficult to make explicit.

In the 42 studies listed and standardised by Brahic and Terreaux (2008), the willingness to pay ranges from less than 1 up to 370 Euros[2] per household per year, with a modal area between 12 and 80 Euros, for ecosystems, habitats or only some of the services that the populations can obtain. This heterogeneity weakens the possibility of taking these studies, even when carefully selected, as a basis for reference values which would be reassuring to find in functional approaches to biodiversity, but we need to look further, in more easily identifiable objects.

## 4.5. Evaluating functional diversity

Understanding biodiversity and ecosystem services in terms of functional biodiversity seems like a more interesting approach to the question of the relationship between biodiversity and services. However, it raises specific difficulties and, in particular, the question of the treatment of the functional redundancy.
To understand the relationship between the state of the biodiversity and the levels of ecosystem services requires a better understanding of the role of the diversity of living beings in ecosystem functions having a more or less direct utility for humans. Scientific knowledge on these interactions is scarce (Carpenter *et al.*, 2006) and related to experimental processes usually concerning only one trophic level (plants, pollinating insects or filtering organisms for example) and a relatively small space scale (Hector *et al.*, 1999; Diaz *et al.*, 2006).

---

[1] Two approaches are possible: the “*hotspot* ”approach, which suggests selecting the sites having greatest diversity, and the “*greedy algorithm* ” approach, which initially selects the site having the strongest biodiversity and then, sequentially, those which have the greatest diversity not yet covered. Polasky and Solow (1999) showed that these two approaches may not lead to the best choice and that this objective involved the possibility of excluding a selected site *a posteriori*. For a summary, see Polasky *et al.* (2005).
[2] This record was set by a study by Mitchell and Carson (1984) concerning and improvement in the water quality of all of the watercourses and lakes in the United States. A study of a set of forests in South Australia reached 280 Euros. These studies seem to have avoided the insertion bias and therefore obtain particularly high values for large ecosystems.

Nevertheless, these experiments have highlighted some interesting results (Diaz *et al.*, 2006; Hector *et al.*, 1999; Loreau *et al.*, 2001, 2003; McCann, 2000; Schwartz *et al.*, 2000). Also, for ecosystems located in temperate zones, there would be a positive relationship between the richness in species and the production of regulation services (resilience faced with disturbances) and provisioning (production of biomass). This relationship, considered marginally, would have a Gaussian curve shape.

The relationships between changes in biodiversity and the production of ecosystem services are in any event very difficult to evaluate because they always relate to the appearance of new interactions between species and not to the species richness or abundance strictly speaking (Yodzis, 1981). Therefore, we cannot consider that there are linear relationships between the change in the size of functional groups and the change in the services to which these groups should in theory relate (Carpenter *et al.*, 2002; McCann, 2000; Diaz *et al.*, 2006). The evaluation of the relationships between the change in the size of the functional groups and the level of the ecosystem services however represents the most promising way of establishing an accounting system for ecosystem services, which takes into account the natural capital at the source of the production of services (Diaz *et al.*, 2006; McNaughton., 1985; MEA, 2005; Schröter *et al.*, 2005).

The monetary evaluation of these biodiversity contributions to the ecosystem functionalities was carried out for the increase in productivity related to the species richness (Costanza *et al.*, 2007), the functional insurance against the uncertainty that accompanies the production of an ecosystem's services (Baumgartner *et al.*, 2007) and the specific interactions which support the correct working of a trophic network (Allen and Loomis, 2006).

In their study of the value of ecological functions in the Changbaisan biosphere reservation (China), Xue and Tisdell (2001) used an approach based on the cost of alternatives (very close to the replacement costs) which led to a valuation of more than 60 million dollars per year, that is to say more than 10 times the value of the yearly production of wood.

Ansink *et al.* (2008) discussed the relevance of evaluating ecosystem services or "ecosystem functions" defined as the capacity of ecosystems to offer services. Their conclusion is that the two approaches have solid conceptual bases and should thus, if carried out with rigour, lead to equal values. Practical considerations often lead analysts to prefer the evaluation of the services and, in the absence of a univocal relationship between services and function, the analyses must be based only on services or only on functions.

Barkman *et al.* (2008) used this argument again and reinforce it, starting from the case of a hydrosystem in Indonesia, noting that to evaluate the functions of ecosystems through the services that the agents can benefit from is the means of circumventing the problem of the non-familiarity of the subjects (in studies based on stated preferences) with the operation of ecosystems, which would lead to a better understanding of the ecological processes.

This analysis leads us to raise a general question: do we have to evaluate the ecosystems according to the services that they provide at a given time or rather according to their "ecological potential" estimated from the analysis of the ecological functions, striving to infer which services could be provided, without underestimating the weight of the constraints related to effective management methods.

## 4.6. Evaluating services

The concept of ecosystem services became an important model for establishing the link between the functioning of ecosystems and human well-being (MEA, 2005). There are a lot of works that propose to evaluate the services rendered by ecosystems, most relating to the recreational value of slightly human adapted areas for various uses, but there are also many other services [1.] The same question is raised by (for example by Arnold and Periz, 2001) or underlies most of the works: is the value of the services provided, possibly cumulated, enough to justify the conservation of a site (or any other natural asset) threatened by the fact that one of the destructive uses to which it is subject (in particular the conversion of the land for agricultural, urban or transport uses) causes it to have a high value[2]?

It is on the basis of the results of a great number of these works that the study published by 13 ecologists, economists and geographers in *Nature* (Costanza *et al.*, 1997) was able to extrapolate and arrive at a value ranging between 1 and 3 times the world gross product for the value of the services offered by all ecosystems. The services taken into account related to the production of food, of various raw materials, recreational uses and water provision, but also to climate regulation and the regulation of atmospheric gases, of the cycle of water, of the formation of soil and control of its erosion, the recycling of nutrients and the purification of effluents.

Even though some services contribute to the production of commercial goods (food production, water provision and production of raw materials are the subject of commercial exchanges)[3], these values are mainly non-commercial. The authors are aware of the great uncertainties that affect their estimates but they however consider that the values are rather low and that more thorough studies are necessary. They also underline that these estimates do not take into account the fact that some of the services would be “literally irreplaceable”.

Constanza *et al.* identify and propose 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.

---

[1] Ecosystem services also contribute to wellbeing in heavily human adapted environments such as urban areas. Bolung and Hunhammar (1999) used the case of the city of Stockholm to show that the survival of urban areas depends on Nature and that cities benefit not only from the ecosystems that surround them but also from “urban ecosystems”.
[2] Within the framework of just commercial uses, the issue of the conversion of arable lands for peri-urban uses or other uses arises. Despite its apparent obviousness the following question can be asked: why is it so advantageous to convert arable lands to build suburban habitats? The answer is deeper than it may appear and the issues are sometimes locally considerable and could become so at larger scales.
[3] However, *a priori*, the prices do not refect the value of ecosystem services but the sum of production costs and rarity rents.

The definition and classification of these services is in fact problematic and has given place to several proposals. Before the *Millennium Assessment* proposed its simplifying system (MEA, 2005), the typology suggested by De Groot *et al.* (2002) had already suggested a quite similar clarification:

1. regulating functions: regulation of the atmosphere, the climate, water flows, water provision, prevention of natural hazards (storms, floods, droughts), soil formation and conservation, recycling of nutrients, liquid waste processing, pollination, biological control;
2. habitat functions: refuge, nurseries;
1. goods and services production function: production of food, of raw materials, of genetic resources, of pharmaceutical resources and of animals and decorative plants;
2. information functions: aesthetics, recreation and (eco) tourism, cultural and artistic, spiritual and historical, scientist and educational inspiration.

For each one of these 23 services, De Groot *et al.* mention, extending the study of Costanza *et al.* whose results they break up, the value ranges for all the ecosystems on the planet. Without considering all these figures again, we should mention 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* (MEA) now constitutes a quite clear and simple framework that it seems reasonable to follow, because it is the result of an unprecedented collective work of comparison and development of consensus. It proposes a typology of ecosystem services in four main categories that have been explained in Chapter IV, and which presents, within a clear framework, all of the relationships between ecosystems and society:

- provisioning (resources, physical services, etc.),
- regulation (water, climate, pollution, diseases, etc),
- culture (recreation, aesthetics, science and education, spiritual, etc),
- support function (primary production, soil formation, etc).

Only the three first can be the subject of an economic evaluation (the support functions are only mentioned as a reminder, since it is a question of maintaining the existing systems, they are valued through the services provided by these systems). For the evaluation methods, refer to the conceptual frameworks likely to be the basis for the practical measurements presented above.

Despite the fact that the MEA is the reference, 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, we can mention the following: 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 (lengthily analysed by Daily, 1997); the complexity of the interactions between the structures, functions and services (Limburg and O'Neill, 2002); the fact that the agents only identify as services those from which they benefit (Boyd and Banzhaf, 2007).

We can quote Loomis *et al.* (2000) who studied five ecosystem services likely to be restored on a 70 km section along the Platte River (waste water dilution, water purification, erosion control, habitat for fish and wildlife, recreational uses). The 100 people questioned accepted that their water bill increased to 21 dollars per month (or 252 dollars per year) for an improvement of these services. By generalising this result to all the households along the river, they obtain a sum of 19 to 70 million dollars (variable according to the interpretation of the null answers); this is a much higher sum than the costs of the conservation projects estimated at 13.4 million to improve these services.

But to review all of the works that have estimated values for one or the other of the services provided by a certain type of ecosystem would be beyond the scope of this work and the reader is encouraged to refer to the main summary studies and meta-analyses mentioned. A set of specific studies will be presented in Chapter VII to justify the development of reference values for a small number of ecosystems.

We must, on the other hand, mention here the general text of Kinzig, Perrings and Scholes (2007), which considers the concept of an ecosystem service as the best perspective complementary to the concept of intrinsic value and argues for the use of the evaluation of ecosystem services as an optimisation mechanism for all investments in conservation by directing them towards where they will be most socially useful. However, they stress the importance, within this framework, of improving our understanding of the relationship between the level of biodiversity and the value of the services.

Thus, we get back to the main reasons that make the evaluation of ecosystem services the best basis for defining reference values aimed at rationalising public choices and the limitations of this approach we are forced to adopt that lead us back to the practical means to institute vigilance. Ecosystem services designate that which is closest to economic situations in our relationships with nature and ecosystems (mobilising means for pursuing an end), but by focusing on the most easily identifiable services we may lose the link with the how living things work and neglect what constitutes the irreplaceable character of Nature, the support of life, and that we try "awkwardly" to designate as an intrinsic value.

## 4.7. The case of the landscapes

The concept of landscape has many meanings. For the ecologist who studies the interactions between the organisation of areas and the ecological processes, the causes and consequences of the heterogeneity of areas, the landscape is an objective concept, a level of analysis. The landscape for the sociologists is a social representation, a subjective concept; for painters and landscape designers, it is an object to be interpreted or built. The European Convention for landscapes retained a phenomenological concept[1] that integrates the interaction between the object and the subject. This seems relevant for evaluations relative to changes. The economic

[1] Article 1 of the CEP: "*Landscape indicates an area of territory as perceived by the population whose character results from the action of natural and/or human factors and their interrelationships…*".

analysis of landscapes emerged recently for analysing an increasing concern about the way space was being occupied by the expansion of cities, proliferation of infrastructures and changes in the rural and urban worlds (Lifran and Oueslati, 2007). It highlighted the economic value of landscape diversity for many economic sectors. It endeavoured to interpret the demand for a greater landscape quality as expressed by individuals and organisations, and tried to specify the nature of the economic value of landscapes.

The value of landscapes results *a priori* from a demand for use by visitors and residents but it also involves option or legacy values, since it is question of conservation. For sites of particular interest, this demand may relate to the entire population of the country (Garrod and Willis, 1995; Willis *et al.*, 2003).

Rambonilaza (2004) listed 28 studies on the evaluation of landscapes, mainly by the contingent method, and analysed the methods of the application of these methods for "landscape benefits", in order to discuss their possibilities of transfer. She showed that there is a complementarity or substitution relationship between the attributes in the landscape demand ("composition effect") which means that we must adapt the contingent method to make it into a "multi-programme" method or make the transition to a *choice experiment,* if we want to obtain significant results within the framework of *ex-ante* evaluation[1]. On the other hand, recourse to the value transfer method seems more suitable for an *ex-post* evaluation. Although the composition effect is not always the subject of an adapted treatment, the results of most of the studies could be transferred. However, the analysis of the results shows that the landscape preferences depend above all on the natural, cultural and social context of the recipients, which makes the transfer of the willingnesses-to-pay from one site to another tricky. Though the meta-analysis of all the existing studies provides a broad spectrum transfer function, its implementation is limited by the low number of original evaluations for multiple situations, from the geographical, ecological and social point of view.

A slightly unexpected result of this work is that it shows that, though the landscape demand is primarily justified by aesthetic and living environment functions, the studies listed reveal the importance of non-use values, related to the aesthetic and ecological functions of the landscapes. The subjects agree to contribute to the maintenance of the landscapes for the satisfaction of their contemporaries or for future generations and sometimes reveal existence values. This report raises a difficulty for the definition of the population to be taken into account, which is already tricky for the integration of the preferences of occasional visitors.

More recently, Cavailhès and Joly (2006) highlighted the complementarity of a geographical approach to the modelling of the landscape as a "scenic volume offered to an observer who looks at everything around him/her" with an analysis by the hedonist price method of more than 4,000 land transactions in Dijon (France). The study shows that the hedonist prices obtained are different in the suburbs of Dijon (where they are often close to zero) and in the peri-urban belt (where forests and agriculture have positive prices while roads have a negative price when these objects

[1] The methods based on stated preferences are *a priori* well suited to measuring values attached to landscape attributes, since they relate to virtual scenarios established according to a controlled combination. Beyond the general limitations of these techniques, in practice they run up against the limited number of attributes that can be used to characterise the changes to landscapes.

are close to residences). Landscape composition in complex or fragmented forms also has a positive price in the peri-urban belt.

The presence of certain biodiversity elements can influence the value of landscapes negatively. In the study of the restoration value of a wetland area in the old marshes of Baux-de-Provence (Westerberg and Lifran, 2008), the expected return of certain species such as mosquitoes influences the willingness-to-pay of the residents negatively and it is the same thing for poplars 1 which would hide the Alpilles landscapes. More generally, elements associated with losses of amenities, well-being or production (crop pests, predators, etc.) are allotted negative values.

The evaluation of the landscapes is structured around two currents which involve both different ideas of landscape and choices of methods. Preferences revealed by hedonist prices are based on an “objectivist” definition of landscape borrowed from geographers (Besançon school, in particular). They aim at establishing correlations between the observable physical characteristics of landscapes in a given place and the price levels on the land and real estate markets. They generally conclude with a weak explanation of the landscape attributes and the crucial role of the nearby landscape (in particular opening characteristics). It is thus difficult to distinguish the effects related to the landscape itself from those of a space search to avoid vicinity effects.

## 4.8. To conclude

As the end of this very incomplete panorama, we can validate the idea, proposed by the Millennium Ecosystem Assessment work that the most realistic approach to the evaluation of ecosystems is by means of an evaluation of the list (organised in the four categories validated by the MEA) of services that users find in these more or less human created environments. The economic analysis thus indicates that the value of an asset can result, at least partially, from the sum of its attributes.

Without calling it into question, we have underlined an important limitation of this approach: it is not certain that the quantity or the quality of the services is very sensitive to the biodiversity of the environments and it is important that, as well as evaluating the services, we maintain forms of vigilance aimed at avoiding the destruction or damaging of environments or species of particular interest, even if they do not yet have an explicit protection status. This prudence leads us back to the debate that we have tried to examine regarding the existence of non-use values, even intrinsic values (the distinction between them remains fuzzy), which may constitute crucial factors in the management or protection choices for these environments or species.

On the other hand, we have said nothing about one simple fact: the services do not necessarily relate to everyone, either because interests diverge, or because practices differ. A striking example is the fact that the biodiversity is, in many sites, in particular when “noticed”, an element of motivation for tourist demand, its value appearing higher for the visitors than for the residents.

[1] Contrary to mosquitos, who have their place in ecosystems, poplars are often judged negatively from an ecological point of view too (exogenic species which tends to make wetlands commonplace).

More largely, Willis *et al.* (2003) highlight the issue of the population for which the values, other than direct use values, must be aggregated. Starting from the question the potential recreational value of Scottish forests, they show that an automatic consequence of the frequent choice of selecting the population of the country as a reference population for potential users or allotting non-use values to ecosystems is to give a much higher value to English forests compared to Scottish forests.

In the last decades, in various contexts, in Europe and elsewhere, the question of payment for environmental services arose (Engel *et al.*, 2008); i.e. a mechanism that establishes a link between the production of value and the implementation of remuneration. The debates are not over but it seems that some elements can be clarified. The first is that a payment is relevant only if it is effective, i.e. if it causes the production of the service to be reinforced. In some cases, it seems more and more that the way to obtain the most value would be to let the ecosystem return to a greater naturalness[1]. A second point is knowing whether there is indeed positive externality or if the people in charge of the actions do not have rights to the assets, which already enable them to appropriate most of the benefits of their actions.

At this stage, going still further would require the scope of this assessment to be extended from scientific knowledge to a set of studies that have attempted to clarify this issue. It would be to deviate from the subject that we are going to pursue by briefly presenting some institutional initiatives which have advanced certain aspects of the evaluation of biodiversity and the services related to ecosystems.

## 5. A recent multiplication of institutional initiatives

The question of the evaluation of natural assets and, more precisely, of ecosystems and biodiversity has become a subject of growing interest for the institutions in charge of economic policies. During recent years, several of these institutions have been led to require studies and summaries on these subjects. Here, we present some examples of the most significant ones.

### 5.1. The building of the reference systems for environmental costs in the transport sector: the work carried out for the State Planning Commission by the Boiteux committee

The State Planning Commission has held discussions regarding the transport sector for more than ten years, one of the objectives of which has been to make evaluations of investment projects more rigorous. It was mainly a question of incorporating standardised money values into the profitability calculations better reflecting the costs and benefits that the community suffers or enjoys because of the new infrastructures.

At the beginning of the years 2000, the ministers in charge of Transport and the Environment also asked the Commission to bring up to date the main conclusions suggested in 1994 in a first report (*Transport: for a better choice of investments*). This

[1] For example, Ohl *et al.* (2008) showed within a rigorous framework that compensation payments to maintain the heterogeneity of habitats are not always legitimate and that effectiveness and equity considerations can lead to over-compensations.

update (*Transport: choice of investments and costs of harmful effects,* 2001) more particularly concerned the evaluation of certain harmful effects such as atmospheric pollution, the greenhouse effect, negative effects on road safety, noise, space congestion in urban areas and cut-off effects caused by large infrastructures. It also covered the highlighting of savings in time, which now have a dominating place in project estimates.

This work required the organisation of a wide consultation and reference to many studies, sometimes divergent, available in France and abroad. The work group, chaired as in 1994 by Marcel Boiteux, honorary president of the EDF, succeeded, despite the uncertainties, in defining a series of precise values usable in the evaluations. It was a question of going, in the fixing of standardised values for non-commercial impacts, as far as allowed by the nature of the subjects covered, the limitations and theoretical and statistical knowledge, and the negotiation in good faith of a consensus between experts and civil servants of different sensitivities.

The work group could not eliminate all of the method and data collection difficulties, but it proposed these first reference systems while waiting for the result of later work, for which it established a priority list. These proposals were audited by the General Civil Engineering Council and led to a revision of the circular governing the economic evaluation methods of large transport infrastructure projects. The work's principles were clearly established.

### *a. Giving a monetary value to the non-commercial advantages and disadvantages of projects*

The monetarisation approach falls under the general concern of giving values to the non-commercial advantages and disadvantages of a project, in order to provide the decision makers with a complete evaluation of the benefits and costs generated by the various operations and alternatives between which they must choose. The report reaffirms the need for socio-economic assessments including evaluations that are as precise as possible of the non-commercial advantages and harmful effects. It distinguishes, on the one hand, what the subject of the commercial evaluation is, based on observed or foreseeable costs and prices, and on the other hand, what the subject of non-commercial evaluations is by means of monetarisation and, finally, what cannot really be attributed a monetary value given the current state of knowledge and manners.

### *b. Establishing a link between the valuation and taxation of the harmful effects*

When it is a question of decisions to be taken in the public sphere, which is usually the case for transport infrastructures, the monetarisation of harmful effects may not just be the subject of instructions for the economic evaluation. The public authorities may also seek to internalise the costs of the harmful effects in the private sphere by means of standards or other devices likely to limit these harmful effects, or even via a tax, which constitutes the most certain way to exactly proportion the incentives to do good. The values suggested in the report may thus be used to calibrate these instruments.

### c. Using standardised values for investment decisions

For reasons of simplicity and equity, we cannot generally stick to calculations adapted to the characteristics of each project. A certain harmonisation, even a standardisation, is desirable. Thus, when the monetarisation of the harmful effect cannot result from the direct confrontation between the polluter and the polluted, we have to refer to values established by the public authorities. The State intervenes to standardise the results of various studies and works and to ensure that all of the stakeholders use the same value, until further notice. The advantage of carrying out economic evaluations of the various infrastructure projects, using identical merit values for all of the projects, accepted by all of the administrations, is obvious.

### d. Integrating the non-commercial advantages and the harmful effects despite uncertainty

The fixing of unit prices for harmful effects, thresholds and calculation methods continually runs up against the fragility and inadequacy of the work mobilised for doing really solid scientific work. The work group did not wish to wait for new work, estimating that it would probably be a very long wait and, if it did this, certain harmful effects would continue to be omitted from assessments – and would thus be counted as zero in calculations due to not knowing which figure to retain – or would be the subject of manipulation by the various interest groups.

Like the formation of prices on the market, the construction of these non-commercial prices must be considered to be a stage in a process of successive errors and corrections. Each stage has its utility. Others will follow this one. It is in this spirit that the values presented in the following tables were worked out by the group.

### e. Working method

The production process for each one of these values was identical: a work group was put in charge of reviewing the literature, collecting together, without *a priori*, everything that might contribute anything to the discussion of the economic values related to this externality (values and methods). This work group presented a summary of this literature at a plenary session and proposed a first justified proposal of value ranges. The consensus was more or less easy depending on the external effects considered, and for some of them, the decision was made after many consultations and discussions, by a majority, the president of the group using his decision-making power as a last resort.

These files, representative of the "state of the art", have fed into the methodological discussions and clarified the fixing of merit values for the monetarisation of the external effects of transport. They also revealed the inevitable limitations of these exercises due:

- to the complexity of the phenomena observed (example: noise), to the many components of the harmful effect studied (economic, physiological, psychological, medical),
- to the heterogeneity of the units and methods of measurement, to the uncertainties of the causal relations,

- to the variability of the impacts according to the geographical or topographic locations,
- to the existence and the use of more or less effective protective devices,
- to the dispersion of the individual and collective assessments of harmful effects according to the living conditions of the stakeholders, to their incomes and to their social or individual characteristics,
- to the heterogeneous national and local situations,
- to surveys and studies favouring one approach or the other.

A proactive approach led to an administrative consensus more or less complete for the recommendation of merit values intended to calculate the socio-economic profitability of the projects, despite the persistence of several obstacles:

- the divergences and weaknesses of the studies available as well as the difficulties of transposing to France the results of studies, often mostly carried out abroad, in particular in the United States,
- the ambiguity of the community's objectives, the declared objective being to better take into account the environmental impact in the socio-economic evaluation of infrastructures, the implicit objective being to revalue environmental cost of transport, in particular roads;
- the practical needs of the calculation which make it necessary to attach unit values to data items that are simple but necessarily reduce the complexity of the impacts and the diversity of individual situations;
- the difficulty in overcoming the apparent contradiction between the fundamental importance of the environmental criteria in the selection and the development of projects and the still minor role of the monetarised environmental impacts in the socio-economic cost-benefit analysis of projects.

Whilst reaffirming the fundamental utility of the socio-economic evaluations obtained by valuing the impacts of any nature that can be quantified and monetarized, in order to be able to compare socio-economic profitabilities, the report several times underlines the need to assess for themselves, backed up by figures along with qualitative analyses, the various terms of the project's utility or disutility that cannot be assigned monetary values. A very deep analysis thus led the work group to not recommend monetary valuation, far too random, for such complex and qualitative subjects as the impact on landscapes, the cut-off effects, space consumption, changes in the assignment of public spaces and congestion, beyond what is currently measurable.

Table V-4: The main values recommended

| Value allotted to human life saved (2000) | | | |
|---|---|---|---|
| | Killed | Seriously injured | Slightly injured |
| Basic value | €1.5 M | €225,000 | €33,000 |

**How to read the table:** *the proposal is that each human life saved by carrying out this project should count as a profit of 1.5 million Euros ascribable to a project. The values in the table will be decreased by a reduction of 33% for road traffic, in order to take into account the fact that the driver assumes part of the risk to which he/she is exposed. The values will grow annually at the same rate as the consumption per capita of households.*

| Value allotted to noise (2000) | | | | | | |
|---|---|---|---|---|---|---|
| Basic calculation | Exposure to sound (in dB) | 55-60 | 60-65 | 65-70 | 70-75 | + than 75 |
| | % depreciation/decibel | 0.4% | 0.8% | 0.9% | 1% | 1.1% |

**How to read the table:** *the proposal is that a monetary value which expresses the damage undergone by the residents should be entered as a cost related to a project. This cost is defined by the depreciation of average rents per square metre of surface occupied and exposed to noise levels exceeding a certain threshold. The selected rent of reference is of 36 francs per square metre and per month (value in 1996). These values will grow annually at the rate of the GDP. These values are also modulated to take into account what the areas concerned are used for (residential, leisure, public corporations) and the different effects on health of the day noise and night noise. The authorised noise level during the day in residential zones is of 60 decibels.*

| Value allotted to a ton of carbon saved (2000) | |
|---|---|
| Price of the ton of carbon (€/tC) | 100 €/tC |

**How to read the table:** *the proposal is that a value of 100 Euros for each ton of carbon saved compared to the reference situation should be entered as a profit ascribable to a project (and as a cost the same value for any ton emitted in addition). This value will grow after 2010 at a rate of 3%.*

| Value allotted to the atmospheric pollution impacts, except for the greenhouse effect (by unit of traffic) (2000) | | | | |
|---|---|---|---|---|
| *2000 Value in €/100.vehicle-km* | Dense Urban | Diffuse Urban | open country | Weighted average |
| Private cars and light commercial vehicles | €2.9 | €1 | €0.1 | €0.9 |
| Heavy trucks | €28.2 | €9.9 | €0.6 | €6.2 |
| Bus | €24.9 | €8.7 | €0.6 | €5.4 |
| *2000 Value in €/100.train-km* | | | | |
| Diesel train (freight) | €458 | €160 | €11 | €100 |
| Diesel train (passengers) | €164 | €57 | €4 | €36 |

**How to read the table:** *the proposal is that each time the carrying out of the project creates – respectively, saves – an additional traffic of 100 vehicles on one kilometre in dense urban environment, a cost – respectively, a profit – of 2.9 Euros (value in 2000) should be entered. This value will grow annually at the same rate as the household consumer expenditure. On the other hand, it will decrease annually by 9.8% to take into account technological progress of the engines.*

| Value allotted to the time-saving for travellers in urban environments (Euros 1998/h) | | |
|---|---|---|
| Type of travel | All France 1998 | Ile-de-France 1998 |
| Professional travel | €10.5/h | €13/h |
| Residence-work travel | €9.5/h | €11.6/h |
| Various travel (shopping, leisure, tourism, etc) | €5.2/h | €6.4/h |
| Average value of all travel | €7.2/h | €8.8/h |

| Value allotted to the time-saving for interurban travellers (Euros 1998/h) | | | | |
|---|---|---|---|---|
| Type | for distances | | Value varying with the distance | Stabilisation for distances over 400 km |
| | < 50 km | < 150 km | | |
| Route | €8.4 /h | - | from €8.4 to 13.7/h | €13.7/h |
| Train 2nd class | - | €10.7 /h | from €10.7 to 12.3/h | €12.3/h |
| Train 1st class. | - | €27.4 /h | from €27.4 to 32.3/h | €32.3/h |
| Air | - | - | €45.7/h | €45.7/h |

**How to read the table**: *the proposal is that an amount of 10.5 Euros (value in 1998) should be entered as a profit ascribable to a project for each hour saved by a user during his/her professional time. For urban travel, this value is contractual. For interurban travel, the value suggested depends on the distance, to take into account the fact that the users are more sensitive to gains obtained on long journeys than on short journeys. This value will grow annually at 70% of the growth rate of household consumption per capita*

| Value allotted to the time-saving for movement of goods | |
|---|---|
| Goods with high value | €0.45/ton/hour |
| Current goods | €0.15/ton/hour |
| Goods with low value | €0.01/ton/hour |

**How to read the table**: *the proposal is that an amount of 0.45 Euros per ton of goods transported and per hour gained should be entered as a profit ascribable to a project to take into account the time saving that goods carriers obtain from the speeding up of the traffic. The values suggested are lower for the goods of lower value. These values will evolve at a rate of two thirds of the evolution of the GDP.*

*The time-saving will also result in a reduction of costs for the owners which will be estimated in constant francs at €31/ hour (1998).*

*Source: CAS, Biodiversity Group*

## 5.2. The creation of the shadow value of carbon: the work carried out at the Strategic Analysis Centre by the Quinet commission

As French engagement in the fight against climate change had been reaffirmed by the environment Grenelle, it was necessary to revise the carbon value of 27 Euros per ton[1] defined by the Boiteux report of 2001[2] and which is currently the reference in the framework recommendations[3] of the Ministry of Ecology and Sustainable Development. This revision must constitute the new consistent reference framework not only for evaluating the socio-economic profitability of large public investments, but also for gauging or evaluating various economic instruments (taxation, subsidies, etc) and regulatory instruments which will be the basis of a public environmental policy.

---

[1] Amount corresponding to a carbon value of 100 Euros per ton: the value of a ton of $CO_2$ is derived from the value of carbon applying a coefficient of 3/11.

[2] This report can be consulted at: www.ladocumentationfrancaise.fr/rapports-publics/014000434/index.shtml.

[3] Framework directive concerning the economic evaluation methods for large transport infrastructure projects of the Ministry of Ecology, Energy, Sustainable Development and Planning (MEEDDAT) dated the 25th of March 2004, updated on the 27th of May 2005.

This frame of reference is thus a medium term signal addressed to all of the public and private players about the price of the carbon that they could face during the next decades.

The commission set up by the CAS at the request of the Prime Minister and chaired by A. Quinet held its discussions in consultation with representatives of the administration, academics and experts, but also with the professional organisations for employers and employees, and environmental protection associations.

### *a. The contextand objectives*

The context of this revision was clearly established.

- *Taking a position within a specified scientific and political framework*

Scientific work 1 relating to climate change has made progress in the understanding of the relationships between human activities, greenhouse gas emissions and the probabilities of an increase in temperatures and climatic disturbances.

The political framework was also specified, with the implementation of greenhouse gas emission reduction commitments, of which some constitute firm international commitments: the Kyoto protocol, which came into effect in 2005, the conclusions of the European Council of March 2007, in which Europe committed to reducing its greenhouse gas emissions by 2020, in France, the programme law fixing the direction of the energy policy[2], which supports the definition of an objective to divide the world greenhouse gas emissions by two between now and 2050.

[1] The publications of the GIEC (Intergovernmental group of experts on the climate) have contributed to refining and circulating scientific and soci-economic expertise on climate.
[2] POPE law data the 13th of July 2005.

- *Taking a position faced with an emerging $CO_2$ market price*

On January the 1st, 2005, Europe set up a European Trading System (ETS) which covers almost 45% of the $CO_2$ emissions coming mainly from the energy sectors and large energy consuming industries. This market resulted in the emergence of a $CO_2$ price, which has fluctuated between 20 and 25 Euros before the financial crisis. Even if this framework could not be used as reference for the discussion, it constituted an unavoidable indication.

- *Benefiting from economic instruments for modelling sustainable development*

Today, due to the progress in economic modelling (models and databases that feed them), we can better represent the evolution of the economies under a “ carbon constraint ”, by taking into account the possibilities of technological changes for each sector as well as the interactions between the value of carbon, the price of fossil energies and global economic balance. These models are thus useful simulation instruments for the debate on the calibration of this reference system.

- *Taking a position in relation to the many works on the monetarisation of carbon in the literature*

### *b. The approach*

Work proceeded in several more or less simultaneous phases:

- A political reflection on the use of this reference system that the report specifies as clearly as possible. The value of carbon retained is intended to be used in the definition of public policies and economic calculations. This value must then be adapted to the specific uses that one would like to make of it, taking into account the economic and financial impacts, the management of professional transitions implied by sectoral changes as well as redistributive effects. The commission takes care to specify that it is proposing a useful reference system but does that it in no way replaces the institutions responsible for defining these policies.
- The production of various simulations based on consensual scenarios by putting several large models in competition to feed the discussion with orders of magnitude and ranges. The report constructed a trajectory compatible with European objectives for 2020 -2050.
- A more theoretical position about the principles and methods in a very polemical academic debate on the subject. The theoretical framework selected was based on a cost/effectiveness approach more consistent with the quantified objectives for the reduction of $CO_2$ emissions. Within this framework, the trajectory of the values of carbon sought must allow the goals to be achieved at the lowest costs. In this sense, the prospect is rather different from that retained in the Stern report, which is based on an analysis of the cost/benefits type. These approaches are complementary but do not answer the same question.
- An analysis of the prices observed on the ETS markets.
- Finally, a systematic comparison of the positions taken by foreign governments, in particular within the Community, American and English.

The carbon reference system thus did not result either from a study or a private individual model, but rather from a collective discussion fed by a state of the art discussed in the light of what the models could simulate, outlined by the theoretical analyses of the problem, taking into account in addition the signal-prices posted on the markets.

Given the uncertainties and degrees of freedom remaining in the recommendations of the economists, the recommended carbon value is thus the fruit of a compromise made within the Commission.
The trajectory of the recommended carbon value is based on three things:

- The value is set at 100 Euros per ton of $CO_2$ by 2030. This value is used as the anchor point for the rest of the analysis. Its relatively high level primarily reflects the ambitious character of the European objectives for the reduction of greenhouse gases and the difficulty in successfully deploying low emission technologies in such a short term.
- After 2030, this value of 100 Euros will grow at the rate of the public discounting rate. This change rule, similar to the Hotelling rule for the optimal exploitation of non renewable resources, is a future conservation rule. It guarantees that the discounted price of a limited resource remains constant over time and is not "crushed" by discounting. A 4%[1] annual growth rate for the value carbon is retained. With these assumptions, the value of carbon increases from 100 Euros per ton of $CO_2$ in 2030 to 200 Euros in 2050.
- From 2010 to 2030, two scenarios were considered:
    - To mechanically observe the Hotelling rule, with a 4% annual discounting rate; which assumes starting from a carbon value of 45 Euros in 2010 to reach 100 Euros in 2030. Such a "jump" would make it possible to integrate a precaution effect that takes into account uncertainties about technological advances and the fact that the cost of damage also depends on the trajectory selected. It would cause two problems however: a problem of consistency over time of the public action, which up to now posted a value of 27 Euros per ton for $CO_2$, and a transition problem, concentrating the change of reference system in only one year, 2010.
    - To start from the Boiteux report value and reach the pivot value of 100 Euros in 2030. This scenario deviates from the Hotelling rule at the beginning of the period, to favour a progressive correction towards the value of 100 Euros in 2030. The transition towards a high carbon price must be gradual to exploit in priority sources of low cost reductions available today and not hold back growth by facilitating the management of economic, social and professional transitions. It is this second scenario that was selected.

    - Table V-5: Shadow value for a ton of $CO_2$ (in Euros 2010)

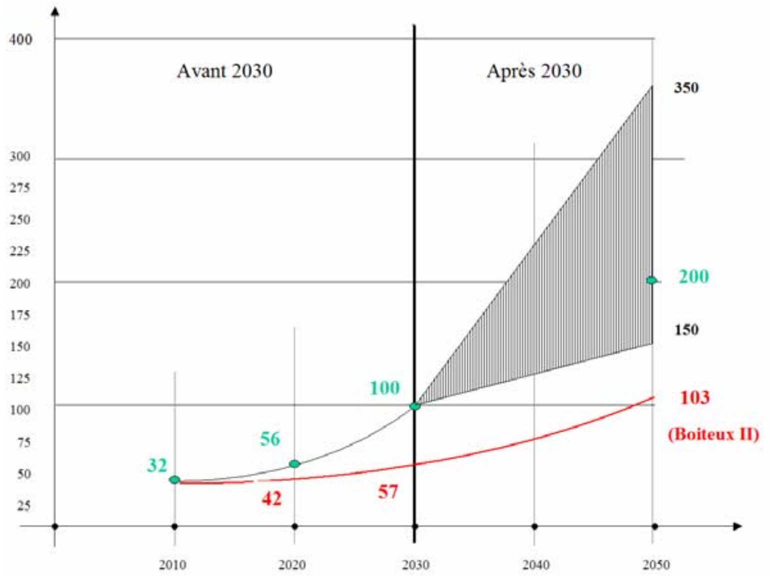
[1] Lebègue D. (2005), *Revision of the discounting rate of public investments*, January.

| | 2010 | 2020 | 2030 | 2050 |
|---|---|---|---|---|
| Recommended value | 32 | 56 | 100 | 200 (150-350) |
| Current value ("Boiteux" Value) | 32(1) | 43 | 58 | 104 |

(1) The Boiteux report gave a value of 27 Euros per ton of $CO_2$ in 2000, corresponding to a value of 32 euros (in euros 2010) after taking into account inflation.

*Source: CAS, Biodiversity Group*

Figure V-6: The carbon reference frame selected




*Source: Centre for strategic analysis*

The recommended carbon values remain surrounded by uncertainties, which become all the larger as the timeline recedes. This is why the 2050 value is framed within a range of 150-350 Euros. This range is aimed at illustrating the extent of the uncertainties that surround the determination of the correct value for carbon beyond 2030, both on the level of the international agreements and on that of the available technologies, whether these are the production systems for non-carbon based energy or the techniques for carbon sequestration and storage. These uncertainties are inherent to any policy to fight against climate change. They will be reduced with the passing of years according to new knowledge.

Revaluation exercises should be held at least every five years. They would be the occasion to give a progress report on the implementation of the reference system suggested and be used to integrate:

- new information on the anticipated cost of the damage, on the cost of the reduction efforts revealed by the permits markets or the price of fossil energies,
- the consequences of a possible variation between the greenhouse gas emissions observed and the target-trajectory aimed at,

- the result of international negotiations, for example the Conference of the Parties planned at the end of 2009 in Copenhagen, if it leads to a new international agreement,
- work of comparable nature on the shadow value of carbon, which could be undertaken at the European level and which would call for a convergence effort between countries. We should remember that a European shadow value still does not exist today.

As the Stern report clearly showed, uncertainty should not lead to inaction. A climatic risk prevention strategy must on the contrary make sure that it:

- uses all information available as well as possible;
- minimises irreversible events. We must act quickly enough to promote the production and spread of new technical solutions and avoid the occurrence of irreversible damage, without taking the risk of "blocking" growth by imposing too harsh constraints on the economy,
- follows a sequential decision-making process, adopting ambitious initial objectives as a precaution.

## 5.3. The cost/benefits analysis approach for conservation projects by the MEEDDAT

The services in charge of economic evaluation at the Ministry for Ecology have, for a few years, been carrying out several cost/benefits analyses in the field of the environment, in particular in regard to the Natura 2000[1] programme and the measures for achieving water purity by 2015[2] (as envisaged by the framework directive on water). These analyses are aimed at comparing the costs of implementing environmental measures with the benefits that society derives from those measures. These costs and benefits may be commercial or non-commercial.

The economic evaluation of a Natura 2000 site was carried out on the plain of Crau, located in Provence-Alpes-Côte d'Azur, which is the last semi-arid steppe of the European continent. The main costs listed in this evaluation are the direct costs, corresponding to the financing of the programme itself (€36/ha/year), as well as the opportunity costs which are the losses of revenue due to the site use restrictions (€24/ha/year). In regard to the benefits, those that were in particular listed were the direct benefits that the sheep stockbreeders and the hay producers of Crau received due to Natura 2000 (€25/ha/year), but especially the social benefits, corresponding to the value that residents of Crau allot to the conservation of the biodiversity of the site. These non-commercial benefits were estimated from the stated preferences of the residents, by using the joint analysis technique. Concretely, a sample of residents was asked to choose between several management programmes for the Crau site, according to the ecological attributes that characterise the programme (species protection, areas helped with organic farming, control of invasive species and conservation of hedges and thickets) and the costs related to carrying this out. It was

[1] "Economic and institutional Evaluation of the Natura 2000 programme: case study on the Crau plain", *Evaluation Letter* of the Directorate for economic studies and environmental evaluation, July 2008.
[2] "Cost/advantage analysis for the restoration of a river: the case of the downstream Gardon ", "Studies and summary" collection, D4E, November 2007.

then estimated that the value of conserving the Crau site for the residents corresponded to their willingness-to-pay to go from a fairly ambitious conservation to an ambitious conservation, namely €66/household/year. By relating this figure to all the households of the seven communes concerned by the designation of the site and to the surface of the site, the social benefit of the protection of Crau was thus evaluated at €182/ha/year. For the Crau plain, the ACA thus shows a positive net profit estimated at €147/ha/year. In other words, the cost of carrying out Natura 2000 on this site is linked to a form of investment in the conservation of biodiversity, answering to a social demand for the protection of nature.

The cost/benefit analysis for achieving water purity relates to a 25 km section of the Gardon river, in the south-east of France. It is a plain river of intermediate size, quite degraded in terms of water quality and artificialisation. Many recreational activities (hiking, fishing, kayaking and bathing) are practiced there. The monetary advantage resulting from the restoration of the Gardon was evaluated by means of the satisfaction of the recreational users and those that the inhabitants of the area would obtain. To measure this increase in well-being, the contingent evaluation method was implemented. Inhabitants of the surrounding communes were surveyed, to get them to state the maximum sum that they would be willing to pay to achieve the good state of this body of water – by explaining simply what the “good state” is and what it will modify. The sum obtained corresponds to the monetary evaluation of the advantage of the restoration of the body of water for recreational uses and the Gardon heritage. The willingnesses-to-pay vary between €14 and 35/household/year according to the following categories of subjects surveyed (non-users, fishermen, hikers, etc.). Again, the total benefit for achieving the good state of the body of water was quantified by applying the values expressed per household to the populations concerned. This benefit thus amounts to €2.86 M/year. The costs of the measures contributing to the achievement of the good state were compared to this benefit – investment cost (the fight against the artificialisation of the Gardon in particular) and operating costs (actions relating to agriculture primarily). The costs and the benefits were then added over a long period – it is indeed a question of achieving a good state, then maintain it. The future values were brought up to their current value, by the discounting process. The result obtained is the difference between the benefits and the discounted costs – it is the net discounted value, expressed in 2010 Euros. The quantified assessment of the cost/benefits analysis is given in table V-6. It shows the economic profitability of the restoration of the lower Gardon since 2010:

- the benefits obtained are higher than the costs,
- the difference between benefits and costs decreases if the actions are deferred.

Table V-6: Net discounted values for various restoration scenarios of lower Gardon

| Evaluated scenario | Net discounted value (difference between discounted benefits and costs) |
|---|---|
| Implementation of DCE actions * | + €38M$_{2010}$ (value for the users),<br>+ €38M$_{2010}$ (heritage value),<br>– €22M$_{2010}$ (cost of the DCE actions),<br>– €18M$_{2010}$ (cost of the actions without DCE),<br>= **€36M$_{2010}$** |

| | |
|---|---|
| Exemption: 6 year carryforward of the deadline (good state in 2021) | + €31M$_{2010}$ (value for the users),<br>+ €31M$_{2010}$ (heritage value),<br>– €20M$_{2010}$ (cost of the DCE actions),<br>– €18M$_{2010}$ (cost of the actions without DCE),<br>= **€24M$_{2010}$** |
| Exemption: 12 year carryforward of the deadline (good state in 2027) | + €26M$_{2010}$ (value for the users),<br>+ €26M$_{2010}$ (patrimonial value),<br>– €18M$_{2010}$ (cost of the DCE actions),<br>– €18M$_{2010}$ (cost of the actions without DCE),<br>= **€16M$_{2010}$** |

(*) DCE: framework directive on water.
*Source: CAS, Biodiversity Group*

## 5.4. OECD Handbook for biodiversity evaluation.

In 2002, after several works about biodiversity management (OCDE, 2001), in particular on incentive mechanisms, the OCDE published a "guide for the decision makers", in other words a summary, written by D. Pearce, D. Moran and D. Miller, aimed at giving a description of the state of the art of the concepts and especially the methods likely to help the economic decision makers to integrate the value of the biodiversity in three types of situations:

- to facilitate the cost/benefits analyses,
- to take into account the environment in the adjustment of the GDP,
- to attach a price to biological resources.

In ten chapters, the work covers most of the questions raised by the evaluation of biodiversity. In particular, it recalls that economic value is an instrumental value and that approaching the total value of the ecosystems other than in the prospect of a progressive degradation is difficult. The evaluation methods are presented and illustrated in three chapters: evaluations based on market prices, evaluations based on stated preferences and the advantage transfer method.

Particular stress is laid on the relationship between the evaluation of biodiversity and decision making, underlining, in particular, the existence of deliberative or participative procedures for obtaining values, their advantages and their disadvantages.

The handbook is completed by a long chapter devoted to the place of evaluation in the development of policies and discusses its advantages and limitations with respect to precautionary approaches. It discusses in particular four criticisms of cost/benefits analysis underlined by Randall (1991):

- the cost/benefits analysis is based on an instrumental concept of the value and the preferences can vary over time; whereas the intrinsic value seems more stable, but difficult to use;
- technological changes have ambiguous effects on the value of biodiversity; they can make it less essential or facilitate its conservation, particularly by lowering the opportunity costs;

- the cost/benefits analysis is "gradual" and can only evaluate limited or gradual changes in biodiversity whose total stock cannot be estimated; it can therefore justify small losses whose effect is however to increase the risk of wider, even total losses;
- the cost/benefits analysis puts the economic theory of the relative value into practice, whereas the value of the biodiversity could be absolute.

These objections to cost/benefits analysis lead the authors to stress the interest of mixed methods combining the instrumental values and the intrinsic values. Cost/benefits analysis would thus be framed by minimum safety standards (in a line close to Randall and Farmer, 1995). The idea would be to combine the cost/benefits analysis of each project with a rule stipulating that the projects and actions of the public authorities must not result in a total deterioration of biodiversity. The handbook concludes with the need for an approach by ecosystems in relation to the measures for protecting charismatic species.

## 5.5. The Economics of Ecosystems and Biodiversity (TEEB)

At the time of the G8+5 meeting of the ministers for the environment organised in Potsdam in March 2007 a joint initiative was launched to draw the attention to the global economic benefits of biodiversity and the cost of the loss of biodiversity and the degradation of ecosystems. The responsibility for this study was entrusted to the economist Pavan Sukhdev, who directs the department of international markets of the Deutsche Bank in Bombay and is the founder of an environmental accountancy project for India.

The TEEB project is organised in two phases: phase 1 is a more forecasting phase on the issues of the erosion of biodiversity, while phase 2 has more operational ambitions both for modelling and calculation. For the moment, only the first phase has been carried out and was the subject of a report presented at the 9th Conference of the Parties, in Bonn in May 2008.

### *a. What does the phase 1 report tells us?*

The objective of this study is not to build alternatives but to provide background information on the current situation and to anticipate what the losses could be by 2050 for:

- the cost of the inaction (but not only that),
- the identification of what are called the vital services (regulation of the local climate, food, access to water, etc.),
- conflicts that will emerge in regard to the access to resources and to ecosystem services (and more particularly vital services),
- the issues for human health because of the erosion of natural resources (sources of molecules and medicines),
- the issues for the poorest who strongly depend on biodiversity for their survival.

In particular, this raises the question of sharing the rent between those who profit from certain services and those who maintain the services taking into account both the maintenance costs and opportunity cost to do it. The report points out the figures in the *Stern Review* which show that 1% of the GDP would be enough to limit the effects of climate change and moves on to biodiversity by insisting on the fact that it constitutes an insurance and that it is in fact a question of maintaining a potential evolution within the meaning used by conservation biologists. The report thus points out the role of information and uncertainty: we must choose the options that are the least destructive, taking into account risks and uncertainties related for example to non-linearities in the impacts on the services and on the discounting rate. This resulted in introducing the concept of option premium in its financial sense: "*The insurance value of biodiversity can be compared with that of the financial markets. following the example of financial values, a varied portfolio of species can be used as a regulating element in view of the fluctuations of the environment or of the market that cause the decline of certain resources. The stabilising effect of a biodiverse portfolio is likely to take on a very particular importance as the environmental change accelerates with the climate change and other impacts of human activities*".
The authors however relativise the approach by services: on the one hand, this value does not represent the full value of the biodiversity and, on the other hand, services may be competitors with each other in the same ecosystem (the question of the interdependence of the services is not taken into account at the present stage). To grasp the various values, the report limits itself to the revealed preference and stated preference methods specifying that "*these methods are convincing but controversial*". It is not make any reference to an approach by costs; but the need for including qualitative approaches and physical indicators is affirmed.

Following this observation, an important passage stresses that the erosion of biodiversity leads to a loss of social well-being (SWB) and that the SWB is not the GDP. Indeed, certain industries will be affected, but others will be able to profit from it. The Net effect on the GDP is thus not clear, at least not in the way that it is currently being calculated.

The phase report announces the ambitions of phase 2 and mentions several tricky points. In the transfer of benefits approach, the value of a service can be found by the extrapolation of the value of the same service to another ecosystem, but the report does not mention any spatial variations of the values. It insists on the other hand on the non-linearity between losses of biodiversity in an ecosystem and losses of services. Lastly, when it measures the value of services, more than 80% of this value is related to carbon storage (a percentage that is all the greater in poor countries). It isn't always easy to know what is being called the value of a service: is this indeed the added-value related to the service or the cost of substitution of the service, which would make it possible to achieve the same level of added-value?

Lastly, a long section analyses the policies and, in particular, the perverse effects that may be opposed to the implementation of actions favourable to biodiversity. To simplify, the report raises the question of the consistency of public policies in relation to production targets and biodiversity conservation objectives (the first often being better defined than the second).

This report thus proposes to improve the existing policies, to work out new policies and to create new markets. The subsidies can be used to encourage the creation of innovations to reduce the harmful effects. It raises the question of substitution between public financing and private financing by giving the example of the landscape bids in Holland. This is a way of setting up mechanisms for financing ecosystem services. In the event of destruction, the “polluter pays” principle should apply, however we also need to promote compensation and creation of new markets with the State's help , without however increasing public expenditure.

Lastly, it proposes that the benefits of conservation should be shared by raising the question of the evaluation of the benefits of conservation and transfer tools, giving two examples: Natura 2000 and tax transfers in Portugal.

It proposes eight key principles for progress:

- the evaluation must stress marginal changes rather than the total value of an ecosystem,
- the evaluation of the services rendered must take into account the context and nature of the ecosystem and its initial state,
- the benefit transfers must be suited to the evaluation of biodiversity (work is to be developed in this direction),
- the values are inevitably guided by the perception of the beneficiaries,
- importance of participative approaches,
- importance of irreversibilities and a better analysis of the resilience phenomena,
- the need to build strong foundations on the biophysical relations to contribute to the evaluation and its credibility,
- the need to develop sensitivity analyses because of uncertainties.

### *b. Issues of phase II*

The phase report underlines four important points that will condition the possible success of the second phase of the study:

- difficulty in avoiding double accounts,
- developing the spatial dimension (little present in the phase report),
- risks analyses and limitation on taking them into account when the changes are not marginal,
- question of the value of stocks from service flow and the role of discounting.

The second phase of the report, under development, involves five work groups which are as follows:

| | | |
|---|---|---|
| Group D0 | Drawing up an evaluation framework, methods, cost analyses. Due date of work. | End of 2009 |
| Group D1 | TEEB and public decision makers | End of 2009 |
| Group D2 | TEEB and public administrations | Start of 2010 |
| Group D3 | TEEB and companies | Mid-2010 |
| Group D4 | TEEB and citizens/ consumers | End of 2010 |

It is thus an extremely ambitious project and an approach that is based on a systemic approach to grasp the extent of the questions raised. The preliminary results seem debatable, in particular in regard to the question of discounting or the evaluation of the service losses compared to a maximum whose bases deserve to be better clarified. They however indicate some really interesting leads based on a thorough analysis of all the scientific knowledge available (see in particular Balmford *et al.*, 2008) and a networked way of working that enables them to call on multiple skills. All the work will be presented at the time of the 10th Conference of the Parties to the Convention on Biological Diversity, in Nagoya in November 2010.

## Conclusions

The economic evaluation of biodiversity and ecosystem services has given rise to a great quantity of work of varied orientations, which endeavoured to propose answers to the multiple difficulties caused by this question. The value of the diversity of living beings is indeed a controversial subject, both on the level of its ethical and social legitimacy and that of its scientific and technical feasibility. This chapter is thus organised into five parts presenting an assessment of the scientific knowledge given (1) the significance of the evaluation, its relationship to decision making and its alternatives; (2) the conceptual framework which is use in the transition from the bases of the value to measurable categories; (3) a review of the methods used to determine empirical measurements; (4) the main results published; up to (5) the presentation of several recent institutional initiatives that, in common with our exercise, weigh up the lessons of economic evaluation in order to improve collective choices in the field of public policies and biodiversity.

To specify the significance of the economic evaluation, we had to reconsider the principles which characterise the economic concept of value and, in particular, anthropocentrism, utilitarianism and marginalism. We thus pointed out some consequences of the instrumental character of this concept, which allots a relative value to the goods and services, according to their contribution to human well-being or, rather, their subjective representation. This value must also reflect the relative scarcity of the services, which is only expressed on a market if institutions and, in particular, the rights allow it. This has led us to reconsider the traditional concepts of externality and public goods, which do not benefit from this mechanism, but also on the "common goods" status of many natural assets and the concept of "merit goods" proposed by Musgrave to indicate goods which, though having an influence on the well-being, are not taken into account in a suitable way in the preferences of the agents.

Then we reconsidered the fundamental distinction between value and price and the bias that the currency can introduce, which enabled us to introduce the question, which is central for this report, of the relationship between evaluation and decision. We underline the basic difference, for the definition of reference values, between values that direct the choices towards a given objective at the lowest costs and must measure the opportunity costs for the projects that we will have to give up to attain it, and the values that aim at the realisation of an optimal conservation objective, as a function of the level of satisfaction of the other objectives of society. The case of biodiversity involves such informational requirements that we limited our ambition to the production of values reflecting the effective interests in the conservation of ecosystems, without explicit reference to a political objective which should also be specified. This part concludes with the presentation of three alternative approaches to economic evaluation (precaution, multicriteria analyses and analyses based on objective measurements) for which we have outlined the advantages and limitations.

The objective of the second part was to specify the adaptations of the conceptual framework of the evaluation necessary for understanding the specificities of biodiversity and ecosystem services. The first point was to clarify the bases of the social value of biodiversity: the ways in which the anthropocentric is a limitation, what the more or less substitutable character of the biodiversity signifies and the fact that biodiversity cannot generally be compared to an economic good but constitutes a property of the ecosystems that can however be evaluated. Conservation of biodiverse ecosystems must in fact be regarded as a mixed public good whose “production” is primarily decentralised, and for which we do not have any clear indication on the level of scarcity and its development.

This good's total economic value amounts to the use values that reflect the services that our societies obtain from ecosystems, and what are known as “non-use” values, related to their heritage status. Several things make this framework complex. Some of the uses are potential and are sometimes presented as a biodiversity insurance value in view of uncertain situations. The economic status of the heritage is more uncertain because it is related both to the existence of altruist preferences (for our contemporaries, future generations or nonhuman species for which we have empathy), but it can also correspond to a civic commitment whose economic evaluation is shaky: is it a question of contributing to the maintenance of common goods or of the simple search for “moral satisfaction”?

The question of time and the discounting these values also raises specific difficulties. Without calling into question the choices retained by the commission Lebègue concerning the discounting rate, the rate at which the ecosystems are becoming rare and the positive income-elasticity that characterises their demand result in recommending increasing relative prices (for example at a rate of 1% per year). The prospect of irreversible and sometimes irreplaceable losses may result in considering a faster increase in certain situations (in particular, for services related to types of ecosystems with significant quantitative regression such as wetlands or perennial meadows, etc.).

The third part reviewed the monetary evaluation methods. We distinguished the methods based on effective costs or on stated preferences which lead to results considered reliable but which only take into account part of the surplus losses related to the degradation of the ecosystems and the methods based on stated preferences that allow *a priori* a more understanding approach to the value of the ecosystems, but whose reliability remains disputed (contingent evaluations) in spite of the promising avenues that situate the subjects questioned in situations closer to the economic choices that are familiar to them.

These techniques are rather demanding to implement and the recommendations aimed at improving reliability are not in the sense of a simplification of the procedures. This fact led to the development of techniques aimed at transferring values from one or more study sites presenting similarities with the sites to be evaluated, without having to develop a heavy protocol. The scientific literature tends to validate the use of these "meta-analyses" provided that they respect transfer protocols that limit the biases. Today, there are databases in which the people responsible can find studies, published or not, from which these transfers of benefits could be made. The still innovative character of these techniques involves reasoned skills, but they are, obviously, something to be developed.

The next part attempted to summarise the information that can be obtained from empirical evaluations of a series of objects related to biodiversity and ecosystems. We distinguished primarily theoretical work aimed at exploring the value of diversity, empirical studies relating to the value of genes in bio prospection, the value of threatened species (but not invading ones), ecosystems or habitats and attempts at relative estimates for functional diversity. This panorama leads us, in line with the work of *Millennium Ecosystem Assessment* whose classification we propose to keep, to validate the idea that a relevant practical approach is to evaluate the services rendered by the ecosystems. We however underline the danger of this approach which can lead to taking little notice of the diversity of the ecosystems: the existence of a linear or refined relationship between the diversity of the ecosystems and the level of the services that can be expected, remains an assumption, supported by the state of the debate on the diversity-stability relationship, but which must be validated on a case-by-case basis and undoubtedly by type of services. We must finally stress that for setting reference values, the question of the reference space for which they must be calculated must also be thought out according to the spatial impact of the projects.

This chapter concludes with the presentation of five institutional initiatives relating to the environmental impact of infrastructures and the evaluation of the biodiversity and of the ecosystem services. There are some others (in particular the committee on the evaluation of services for water and terrestrial ecosystems which depend on these, created by the American National Research Council, see Heal, 2005), however we limited ourselves to cases that are specially enlightening for the work of our group. In each case, we tried to provide the main things that can be used to establish the link between the means and methods implemented and the objectives pursued by these groups.

Given the state of the scientific knowledge and these experiments, can we think about producing "reference values" with wide spatiotemporal validity? This would undoubtedly be too ambitious an objective, but we could start with partial results with a more limited spectrum of application, but nevertheless having a practical use and, thus, making incremental progress which will be validated by experience.

# Bibliographic References

Aldred J. (2006), « Incommensurability and monetary valuation ». *Land Economics*, 82 (2), 141-161.

Alexander A. M., List J. A., Margolis M. et d'Arge R. C. (1998), « A method for valuing global ecosystem services », *Ecological Economics*, 27, 161-170.

Allen B. P. et Loomis J. B. (2006), « Deriving values for the ecological support function of wildlife: an indirect valuation approach », *Ecological Economics*, 56, 49-57.

Allsopp M. H., de Lange W. J. et Veldtman R. (2008), « Valuing insect pollination services with cost of replacement », *PLoS ONE*, 3 (9) (September), www.plosone.org e3128.

Alvarez-Farizo B. et Hanley N. (2006), « Improving the process of valuing non-market benefits: combining citizens' juries with choice modelling », *Land Economic*, 82, 465–478.

Amigues J.-P. et Desaigues B. (2001), « A cost-benefit analysis of biodiversity conservation programmes in the Garonne valley: French case study on biodiversity incentive measures », *in Case Studies for the Handbook of Biodiversity Valuation: A Guide for Policy Makers*, OCDE, Paris.

Amigues J.-P., Desaigues B. et Vuong Q. (1996), « L'évaluation contingente: controverses et perspectives, *Cahiers d'économie et sociologie rurales*, (Contingent Assessment, Controversies and Prospects) n° 39-40), p. 124-150.

Amigues J.-P., Boulatoff C., Desaigues B., Gauthier C. et Keith J. E. (2002), « The benefits and costs of riparian analysis habitat preservation: a willingness to accept/willingness to pay contingent valuation approach », *Ecological Economics*, 43(1): 17-31.

Ando A., Camm J., Polasky S. et Solow A. (1998), « Species distributions, land values, and efficient conservation », *Science*, 279, 2126-2128.

Anielski M. et Wilson S. (2005), *Counting Canada's Natural Capital: Assesing the Real Value of Canada's Boreal Ecosystems,* Canadian Boreal Initiative and Pembina Institute, Ottawa, Canada, 79 p.

Ansink E., Hein L. et Hasund K P. (2008), « To value functions or services? An analysis of ecosystem valuation approaches », *Environmental Values*, Vol. 17, n° 4, November, 489-503.

Armsworth P. R, Kendall B. E. et Davis F. W. (2000), « An introduction to biodiversity concepts for environmental economists », *Resource and Energy Economics*, 26(2), 115-134.

Arnold J. E. M. et Periz M. R. (2001), « Can non-timber forest products match tropical forest conservation and development objectives », *Ecological Economics*, 39, 437-447.

Arrow K.J. (1951), *Social Choice and Individual Values,* New Haven, Yale University Press, 124 p.

Arrow K. J. et Fisher A. C. (1974), « Preservation, uncertainty and irreversibility », *Quarterly Journal of Economics*, 87, 312-319.

Arrow K. J., Solow R., Portney P. R., Learner E. E., Radner R. et Shuman H. (1993), *Report of the NOAA Panel on Contingent Valuation*, Federal Register 58(10): 4602-4614.

Arrow K. J., Bolin B., Costanza R., Dasgupta P., Folke C., Holling C. S., Jansson B. O., Levin S., Mäler K G., Perrings C. et Pimentel D. (1995), « Economic growth, carrying capacity and the environment », *Science*, 268, 520-521.

Ashworth L., Quesada M., Casas A., Aguilar R. et Oyama K. (2009), « Pollinator-dependent food production in Mexico », *Biological Conservation* (2009), doi:10.1016/j.biocon.2009.01.016

Aulong S., Erdlenbruch K. et Figuières C. (2005), « Un tour d'horizon des critères d'évaluation de la diversité biologique », *Économie Publique*, *(A Survey Of the Evaluation Criteria of Biological Diversity)* n° 16 (2005/1), p. 3-46.

Balmford A., Bruner A., Cooper P., Costanza R., Farber S., Green R. E., Jenkins M., Jefferiss P., Jessamy V., Madden J., Munro K., Myers N., Naeem S., Paavola J., Rayment M., Rosendo S., Roughgarden J., Trumper K. et Turner R. K. (2002), « Economic reasons for conserving wild nature », *Science*, 297, 950-953.

Balmford A., Gaston K., Blyth S., James A., et Kapos V. (2003), « Global variations in terrestrial conservation costs, conservation benefits, and unmet conservation needs », *Proceeding of the Natinal Academy of Sciences 100*, 104-1050.

Balmford A., Rodrigues A. S. L., Walpole M., ten Brink P., Kettunen M., Braat L. et de Groot R. (2008), *The Economics of Biodiversity and Ecosystems: Scoping the Science.*, Cambridge, UK: European Commission, September, 300 p.

Barbier E. B. (2000), « Valuing environment as an input: review of applications to mangrove-fisherie linkage », *Ecological Economics*, 35, 47-61.

Barbier E. B., Burgess J. C. et Folke C. (1994), *Paradise Lost? The Ecological Economics of Biodiversity*, Earthscan, London.

Barde J.-P. et Pearce D. W. (1991), *Valuing the Environment: Six Case Studies*, Earthscan, London.

Bark R. H., Osgood D. E., Colby B. G., Katz G. et Stromberg J. (2009), « Habitat preservation and restauration: Do homebuyers have preferences for quality habitat? », *Ecological Economics* (2008), doi:10.1016/j.ecolecon.2008.10.005.

Barkmann J., Glenk K., Keil A., Leemhuis C., Dietrich N., Gerold G. et Margraaf R. (2008), « Confronting unfamiliarity with ecosystem function: The case for an ecosystem service approach to environmental valuation with stated preference method », *Ecological Economics*, 65, 48-62.

Bateman I. J. (2008), « Valuing "X" (when "X" is uknown). Key note lecture », *10th BioEcon workshop*, Cambridge (UK), September 29-30th.

Bateman I. et Willis K. G., (eds) (1999), *Valuing Environmental Preferences: Theory and Practice of the Contingent Valuation Method in the U.S, E.C. and Developing Countries*, Oxford: Oxford University Press.

Bateman I. J. et Jones A. P. (2003), « Contrasting conventional with multi-level modelling approaches to meta-analysis: Expectation consistency in U.K. woodland recreation values », *Land Economics*, 79 (2): 235-258.

Baumgärtner S. (2007), « The insurance value of biodiversity in the provision of ecosystem services », *Natural Resource Modeling*, 20 (1), Spring, 87-104.

Baumgärtner S., Becker C., Faber M. et Manstetten R. (2006), « Relative and absolute scarcity of nature. Assessing the roles of economics and ecology for biodiversity conservation », *Ecological Economics*.

Baumgärtner S. et Quaas M. F. (2007), « Agro-biodiversity as natural insurance and the development of financial insurance markets », *Discussion Paper*, Department of Economics, University of Heidelberg, September.

Baumgärtner S. et Quaas M. F. (2005), « The private and public insurance value of conservative biodiversity management », *Discussion Paper*, Department of Economics, University of Heidelberg, December.

Beaumais O., Laroutis D. et Chakir R. (2008), « Conservation *versus* conversion des zones humides: Une analyse comparative appliquée à l'estuaire de la Seine », *Revue d'Économie Régionale et Urbaine*, (Conservation versus Conversion of Wetland: a Comparative Analysis Applied to the Seine River Estuary) n° 3/2008.

Beckermann W. (1994), « 'Sustainable environment'. Is it a useful concept? », *Environmental Values*, 3, 191-209.

Béné C. et Doyen L. (2008), « Contribution values of biodiversity to ecosystem performances: A viability approach », *Ecological Economics*, 68 (1-2), 14-23.

Bennett J. et Blamey R. (eds) (2001), *The Choice Modelling Approach to Environmental Valuation*, Edward Elgar, Cheltenham.

Bergstrom J. C. et Taylor L. O. (2006), « Using meta-analysis for benefits transfer: Theory and practice », *Ecological Economics*, 60 (2), December, 351-360.

Berrens R. P., Brookshire D. S., McKee M. et Schmidt C. (1998), « Implementing the safe minimum standard approach: two case studies from the U.S. Endangered Species Act », *Land Economics*, May.

Bervoets S. et Gravel N. (2007), « Appraising diversity with an ordinal notion of similarity: an axiomatic approach », *Mathematical Social Sciences*, 53, 259-273.

Bin O. et Polasky S. (2005), « Evidence on the amenity value of wetland in rural settings », *Journal of Agriculture and Applied Economics*, 37(3), 589-602.

Bishop R. C. (1978), « Endangered species and uncertainty: The economics of a safe minimum standard », *American Journal of Agricultural Economics*, 60(1): 10-18.

Bockstael N. E., Freeman A. M., Kopp R. J., Portney P. R. et Smith V. K. (2000), « On measuring economic value for nature », *Environmental Science and technology*, 34, 1384-1389.

Boman M. (2008), « To pay or not to pay for biodiversity in forests – what scale determines responses to willingness to pay questions with uncertain response options? », *Journal of Forest Economics*, 15, 79-91.

Bonnieux F. et Desaigues B. (1998), *Économie et politique de l'environnement*,(Environment Economics and Policy) Dalloz, Paris, 250 p.

Bonnieux F. et Legoffe P. (1997), « Valuing the benefits of landscape restauration: A case study of the Cotentin in Lower-Normandy, France », *Journal of Environment Management*, 50 (3), 321-333.

Bonnieux F. et Rainelli P. (1990), *Évaluation économique et catastrophe écologique*, (Economical Assessment and Ecological Disaster) Paris, Economica.

Bosetti V., Conrad J. M. et Messinat E. (2004), « The value of flexibility: Preservation, remediation, or development for Ginostra? », *Environmental and Resource Economics*, 29 (2), October, 219-229.

Boyd J. et Banzhaf S. (2007), « What are ecosystem services? The need for standardized environmental accounting units », *Ecological Economics*, 63 (2–3), 616–626.

Boxall P. C., Adamowicz W. L., Swait J., Williams M. et Louviere J. (1996), « A comparison of stated preference method for environmental valuation », *Ecological Economics*, 18, 243-253.

Braat L. et ten Brink P. (2008), *The Cost of Policy Inaction (COPI) - The Case of not Meeting the 2010 Biodiversity Target*, Report to the European Commission, May 29.

Brahic E., Clément V., Moureau N. et Vidal M. (2008), *À la recherche des Merit Goods*, (Looking for Merit Goods) LAMETA, DR n° 2008-08, avril, 20 p.

Brahic E. et Terreaux J.-P. (2009), *Évaluation économique de la valeur de la biodiversité en forêt. Guide méthodologique*, (Economical Assessment of the Biodiversity Value in Forests. Methodological Guide) CEMAGREF-MAP-MEEDDAT-ONF, 350 p.

Brander L. M., Raymond J. G., Florax M. et Vermaat J. E. (2006), « The Empirics of wetland valuation: A comprehensive summary and meta-analysis of the literature », *Environmental and Resources Economics*, 33, 223–250.

Brock W. A. et Xepapadeas A. (2003), « Valuing biodiversity from an economic perspective: A unified economic, ecological, and genetic approach », *American Economic Review 93* (5), 1597-1614.

Brouwer R. (2000), « Environmental value transfer: state of the art and future prospects », *Ecological Economics*, 32: 137–152.

Brouwer R., Langford I.H., Bateman I. J. et Turner R. K. (1999), « A meta-analysis of wetland contingent valuation studies », *Regional Environmental Change*, 1 (1), November, 47-57.

Brown G. M. et Shogren J. F. (1998), « Economics of the endangered species act », *Journal of Economic Perspectives*, 12 (3), 3-20.

Brown T. C. et Slovic P. (1988), « Effects of context on economic measures of value », *in* G. L.. Peterson, B. L. Driver et R. Gregory (eds), *Amenity resource valuation: Integrating economics with other disciplines*, State College, PA: Venture, p. 23-30.

Carpenter S. R. et Brock W. A. (2008), « Adaptive capacity and traps », *Ecology and Society*, 13(2): 40.

Carpenter S. R., Brock W. A. et Ludwig D. (2002), « Collapse, learning, and renewal », *in* L. H. Gunderson et C. S. Holling, (eds), *Panarchy. Understanding Transformations in Human and Natural Systems*, Washington D.C., Island Press, 173-193.

Carpenter S. R., DeFries R., Dietz T., Mooney H. A., Polasky S., Reid W. et Scholes R. J. (2006), « Millennium Ecosystem Assessment: research needs », *Science*, 314, 257-258.

Carson R. T., Hanemann W. M., Kopp R. J., Krosnick J. A., Mitchell R. C., Presser S., Ruud P. A. et Smith V. K. (1996), « Was the NOAA panel correct about contingent valuation? », *Discussion Paper* n° 96-02, Resources for the Future, Washington D.C.

Carson R. T., Hanemann W.M., Kopp R. J., Krosnick J. A., Mitchell R. C., Presser S., Ruud P. A. et Smith V. K. (1995), « Temporal reliability of estimates from contingent valuation », *Discussion Paper* n° 95-37, Resources for the Future, Washington D.C.

Cavailhès J., Brossard T., Hilal M., Joly D. et Tourneux P. F. (2007), « Le prix des paysages périurbains »(The Cost of Peri-Urban Landscapes) (, *Économie Rurale*, n° 297-298, janvier-mars.

Cavailhès J. et Joly D. (éd.) (2006), *Les paysages périurbains et leur prix*,(Peri-Urban Landscapes and their Cost) Besançon, Presses Universitaires de Franche-Comté, 192 p.

Centre d'analyse stratégique (2008a), « La valeur du vivant: quelle mesure pour la biodiversité ? (The Value of Living Species: Which Measure for Biodiversity ?)» *La Note de veille*, n° 89, février.

Centre d'analyse stratégique (2008b), *La valeur tutélaire du carbone (Tutelary Value of Carbon),* Rapport de la commission présidée par A. Quinet, coll. Rapports et documents, n° 16, La Documentation française, juin.

Chambers C. M. et Whitehead J. C. (2003), « A contingent valuation estimate of the benefits of wolves in Minnesota », *Environmental and Resource Economics*, 26(2): 249-267.

Chavas J. P. (2009), « On the productive value of biodiversity », *Environmental and Resource Economics*, 42 (1), 109–131.

Chichilniski G. et Heal G. (1998), « Economic returns from the biosphere », *Natur*, 391, February, 629-630.

Christie M., Hanley N., Warren J., Murphy K., Wright R. et Hyde T. (2006), « Valuing the diversity of biodiversity », *Ecological Economics*, 58 (4), 304–317.

Clarke H. R. et Reed W. J. (1990), « Land development and wilderness conservation policies under uncertainties: A synthesis », *Natural Resource Modelling*, 4 (1), 11-37.

Colson F. et Stenger-Letheu A. (1996), « Évaluation contingente et paysages agricoles. Application au bocage de Loire-Atlantique »,(Contingent Assessment and Agricultural Landscapes: Application to the Loire-Atlantique Bocage)*Cahiers d'économie et sociologie rurales*, n° 39/40, p. 151-177.

Conrad J. M. (2000), « Wilderness: options to preserve, extract or develop », *Resource and Energy Economics 22*, 205-219.

Conrad J. M. (1997), « On the option value of old-growth forest », *Ecological Economics*, 22, 97-102.

Convention on Biological Diversity (2007), « An exploration of tools and methodologies for valuation of biodiversity and biodiversity resources and functions », *CDB Technical Series*, n° 28.

Convention on Biological Diversity (2001), « The value of forest ecosystems », *CDB Technical Series*, n° 4.

Costanza R. (2008), « Ecosystem services: multiple classification systems are needed », *Biological Conservation*, 141, 350-352.

Costanza R., Fisher B., Mulder K., Liu S. et Christopher T. (2007), « Biodiversity and ecosystem services: a multi-scale empirical study of the relationship between species richness and net primary production », *Ecological Economics*, 61, 478–491.

Costanza R. et Wilson M. A. (2002), « Economic and ecological concepts for valuing ecosystem services », *Ecological Economics*, 41, 375-392.

Costanza R., d'Arge R., de Groot R., Farber S., Grasso M., Hannon B., Limburg K., Naem S., O'Neil R., Paruelo J., Raskin R., Sutton P. et van der Belt M. (1997), « The value of the world ecosystems and natural capital », *Nature*, 387, 253-260.

Costello C. et Ward M. (2006), « Search, bioprospecting and biodiversity conservation », *Journal of Environmental Economics and Management*, 52, 615-626.

Craft A. R. et Simpson R. D. (2001), « The value of biodiversity in pharmaceutical research with differentiated products », *Environmental and Resource Economics*, 18 (1), 1-17.

Crocker T. (1985), « On the value of the condition of a forest stock ». *Land Economics,* 61, 244-254.

Cropper M. L. et Oates W. (1992), « Environmental economics: a survey », *Journal of Economic Literature*, 30, 675-740.

D4E (2008), « La prise en compte de la biodiversité dans les études d'impacts des grands projets d'infrastructures de transport », ("Taking account of Biodiversity in Studies Relating to the Impact of Major Projects for Transportation Infrastructures") *La Lettre de la Direction des études économiques et de l'évaluation environnementale*, n° 13, juillet.

Dachary-Bernard J. (2007), « La méthode des choix multi-attributs appliquée aux Monts d'Arrée »( The Multi-Attribute Choice Method Applied to the Monts d'Arrée), *Cahiers d'économie et sociologie rurales*, n° 84-85, p. 133-166.

Dachary-Bernard J. (2005), « Une évaluation économique du paysage. Une application de la méthode des choix multi-attributs aux Monts d'Arrée »(An Economical Landscape Assessment. An Application of the Attribute Choice Method Applied to the Monts d'Arrée, *Économie et Statistiques*, n° 373, p. 57-74.

Daily G. C. (1997), *Nature's Services. Societal Dependence on Natural Ecosystems,* Island Press, Washington DC.

Daly H. E. et Townsend K. N. (1993), *Valuing the Earth: Economics, Ecology, Ethics*, Cambridge, Mass., MIT Press, 387 p.

Daly H. (1995), « On Wilfred Beckerman's critique of sustainable development », *Environmental Values*, 4 (1), February, 49-70.

Dasgupta P. (2001), *Human Well-Being and the Natural Environment*, Oxford University Press, Oxford, 305 p.

Dasgupta P. et K.-G. Mäler (2003), « The economics of non-convex ecosystems: Introduction », *Environmental and Resource Economics*, 26 (4), December, 499-525.

Davis R. K. (1963), *The Value of Outdoor Recreation: An Economic Studies of the Maine Woods*, Ph. D. dissertation, Harvard University, Cambridge, MA.

Desaigues B. et Ami D. (2001), « An estimation of the social benefits of preserving biodiversity », *International Journal of Global Environmental Issues*, 1 (1), 73-86.

Desaigues B. et Point P. (1993), *Économie du patrimoine naturel: la valorisation des bénéfices de protection de l'environnement (Economics of Natural Patrimony: Valuing Environment Protection Benefits)*, Paris, Economica, 317 p.

Desvouges W. H., Johnson F. R., Dunford R. W., Hudson S. P. et Wilson K. N. (1993), « Measuring natural resource damage with contingent valuation: test of validity and reliability », *in* Hausman J. A. (ed.), *Contingent Valuation: A Critical Assessment*, Amsterdam, North-Holland.

Diamond J. (2006), *Effondrement. Comment les sociétés décident de leur disparition ou de leur survie*, (Collapsing. How Societies Decide if they will Survive or Go Extinct) Gallimard (*NRF Essais*), Paris, 648 p. (English publ., 2005).

Diamond J. (2000), *De l'inégalité parmi les sociétés. Essai sur l'homme et l'environnement dans l'histoire*, Gallimard (About Inequalities Among Societies. An Essay on Human Beings and on Environment in History)(*NRF Essais*), Paris, 484 p. (English publ., 1997).

Diamond A. et Hausman J. A. (1994), « Contingent valuation: Is some number better than no number? », *Journal of Economic Perspective*, 8 (4), 45-64.

Diamond A. et Hausman J. A. (1993), « On contingent valuation measurements of nonuse values », *in* Hausman J. A. (ed.), *Contingent Valuation: A Critical Assessment*, Amsterdam, North-Holland.

Díaz S., Fargione J., Chapin III F. S. et Tilman D. (2006), « Biodiversity loss threatens human well-being », *PLOS Biology*, 6 (3), 300-1305.

Drechsler M. et Wätzold F. (2009), « Applying tradable permits to biodiversity conservation: Effects of space dependent conservation benefits and cost heterogeneity on habitat location », *Ecological Economics*, 68 (4), 1083-1092.

Eiswerth M. E. et Haney J. C. (2001), « Maximizing conserved biodiversity: why thresholds matter ». *Ecological Economics,* 38, 259-274.

Ekins P., Simon S., Deutsch L., Folke C. et De Groot R. (2003), « A framework for the practical application of the concepts of critical natural capital and strong sustainability », *Ecological Economics*, 44 (2-3), 165-185.

Elster J. (1989), « Social norms and economic theory », *Journal of Economic Perspectives*, 3, 99-117.

Elster J. (1979), *Ulysses and the Sirens: Studies in Rationality and Irrationality*, Cambridge University Press, Cambridge, UK.

Engel S., Pagiola S. et Wunder S. (2008), « Designing payments for environmental services in theory and practice: An overview of the issue », *Ecological Economics*, 65, 663-674.

Enjolras G. et Boisson J.-M (2007), « Valuing lagoons using a meta analytIcal approach: methodological and practical issues », *Paper* presented at the 15th EAERE Annual Conference, Thessaloniki, Greece, June 27-30.

Farber S. C., Costanza R. et Wilson M. A. (2002), « Economic and ecological concepts for valuing ecosystem services », *Ecological Economics*, 41, 375-392.

Faucheux S. et O'Connor M. (eds) (1997), *Valuation for Sustainable Development: Methods and Policy Indicators*. Edward Elgar, Aldershot.

Fiala N. (2008). « Measuring sustainability: Why the ecological footprint is bad economics and bad environmental science », *Ecological Economics*, 67, 519-525.

Fiorito R. et Kollintzas T. (2004), « Public goods, merit goods and the relation between private and government consumption », *European Economic Review,* 48, 1367-98.

Fisher A. C. et Hanemann W. M. (1986), « Option value and the extinction of species », *in* V. K. Smith (ed.), *Advances in Applied Microeconomics*, Vol. 4, Greenwich, Connecticut: JAI Press, 169-190.

Fisher B., Turner R. K. et Morling P. (2009), « Defining and classifying ecosystem services for decision making », *Ecological Economics*, 68 (2009) 643–653.

Foley J. A., Monfreda C., Ramankutty N. et Zaks D. (2007), « Our share of the planetary pie », *PNAS*, 104 (31), July 31, 12585-12586.

Fromm O. (2000), « Ecological structure and functions of biodiversity as elements of its total economic value », *Environmental and Resource Economics*, 16, 303-328.

Gallai N., Salles J.-M., Settele J. et Vaissière B. (2009), « Economic valuation of the vulnerability of world agriculture confronted with pollinator decline », *Ecological Economics*, 68 (1), 810–821.

Garrod G. D. et Willis K. G. (1995), « Valuing the benefits of the South downs environmentally sensitive area », *Journal of Agricultural Economics*, 46, 160,173.

Géniaux G. (2002), « Le *Mitigation Banking*: un mécanisme décentralisé au service des politiques de "*no net loss* "»(A Decentralised Mechanism to Serve "No Net Loss" Policies), Actes et communications de l'INRA, 17.

Géniaux G. (1999), *Évaluation et régulation des impacts environnementaux*, (Assessment and Regulation of Environmental Impacts) thèse de doctorat es sciences économiques, université de la Méditerranée, janvier, 306 p.

Ghazoul J. (2007), « Recognizing the complexities of ecosystem management and the ecosystem service concept », *Gaia*, *16* (3), 215-221.

Godard O. (2005), « Les conditions d'une gestion économique de la biodiversité - Un parallèle avec le changement climatique »(The Conditions for an Economical Management of Biodiversity - A cg

Godard O. (2004), « De la pluralité des ordres – Les problèmes d'environnement et de développement durable à la lumière de la théorie de la justification »,(The Environment and Sustainable Development Problems in the Light of the Justification Theory) *Géographie Économie Société*, 2004/3 (6), p. 303-330.

Goeschl T. et Swanson T. (2002), « The social value of biodiversity for R&D », *Environmental and Resource Economics*, 22, 477–504.

Gollier C., Jullien B. et Treich N. (2000), « Scientific progress and irreversibility: an economic interpretation of the "Precautionary Principle" », *Journal of Public Economics*, 75, 229–253.

Goulder L. H. et Donald K. (1997), « Valuing ecosystem services: philosophical bases and empirical method", *in* Daily G.C. (ed.), *Nature's Services: Societal Dependance upon Natural Ecosystems*, Island Press, Washington D.C., 23-48.

Gouyon P.-H. (1994), « La biodiversité dans une perspective historique », (Biodiversity in a Historical Perspective) *Courrier de l'environnement*, n° 23, INRA.

Gouyon P.-H. (2001), *Les harmonies de la nature à l'épreuve de la biologie*, (Harmonious settings of Nature Confronted with Biology) (The Nature Paris, INRA (« Sciences en Questions »).

Green I.-M., Folke C., Turner R. K. et Bateman I. J. (1994), « Primary and secondary values of wetland ecosystems », *Environmental and Resource Economics*, 4 (4), 55-74.

De Groot R. S., Wilson M. A. et Boumans R. M. J. (2002), « A typology for the classification, description and valuation of ecosystem functions, goods and services », *Ecological Economics*, 41, 393–408.

Haberl H. *et al.* (2009), « Toward an integrated model of socioeconomic biodiversity drivers, pressures and impacts. A feasibility study based on three long-term socio-ecological research platforms », *Ecological Economics* (2009), doi:10.1016/j.ecolecon.2008.11.013.

Haberl H., Erb K. H., Kausmann F., Gaube V., Bondeau A., Plutzar C., Gingrich S., Lucht W. et Fischer-Kowalski M. (2007), « Quantifying and mapping the human appropriation of the net primary production in earth's terrestrial ecosystems », *PNAS*, 104 *(31)*, 12942-12947.

Hamaide B., ReVelle C. S. et Malcolm S.A. (2006), « Biological reserves, rare species and the trade-off between species abundance and species diversity », *Ecological Economics*, 56 (4), 570-583.

Hanemann M. (1991), « Willingness to pay and willingness to accept: how much can they differ? », *American Economic Review*, 81, 635-647.

Hanemann W. M. (1989), « Information and the concept of option value », *Journal of Environmental Economics and Management*, 16 (1), January, 23-37.

Hanemann W. M. (1988), « The value of biodiversity », *in* Wilson E. O. (ed.), *Biodiversity*, The National Academies Press, Washington DC.

Hanley N. (1998), « Resilience in social and economic systems: a concept that fails the cost benefit test? », *Environment and Development Economics*, 3 (2), May, 221-262.

Hanley N., MacMillan D., Wright R. E., Bullock C., Simpson I., Parsisson D. et Crabtree B. (1998), « Contingent valuation versus choice experiments: estimating the benefits of environmentally sensitive areas in Scotland », *Journal of Agricultural Economics*, 49 (1), 1-15.

Hanley N., Spash C. et Walker L. (1995), « Problems in valuing the benefits of biodiversity protection », *Environmental and Resource Economics*, 5 (3), 249-272.

Hanley N. et Shogren J. F. (2002), « Awkward choices: economics and nature conservation », *in* Bromley D. W. et Paavola J. (eds), *Economics, Ethics and Environmental Policy: Contested Choices*, Blackwell Publishing, Oxford.

Hanley N., Ready R., Colombo S., Watson F., Stewart M. et Bergmann E. A. (2008), « The impacts of knowledge of the past on preferences for future landscape change », *Working Paper*, Division of Economics, University of Stirling, 44 p.

Hanley N., Willis K., Powe N. et Anderson M. (2002), *Valuing the Benefits of Biodiversity in Forests*, Report to the Forestry Commission, Edinburgh, 66 p.

Hanley N., Schlapfer F. et Spurgeon J. (2003), « Aggregating the benefits of environmental improvements: distance decay functions for use and non-use values », *Journal of Environmental Management*, 68 (3), 297-304.

Hardin G. (1968), « The Tragedy of the Commons », *Science*, 162, 1243-1248.

Hausman J. A. (ed.) (1993), *Contingent Valuation: A Critical Assessment*, Amsterdam, North-Holland.

Heal G. (ed.) (2005), *Valuing Ecosystem Services: Toward Better Environmental Decision-Making*, Report of the National Research Council, National Academy Press, Washington DC, 278 p.

Heal G. (2004), « Economics of biodiversity: An introduction ». *Resource and Energy Economics 26*, 105-114.

Heal G. (2000a), *Nature and the Marketplace: Capturing the Value of Ecosystem Services*, Island Press, Washington DC.

Heal G. (2000b), « Biodiversity as a commodity », *in Encyclopedia of Biodiversity*,. Academic Press.

Heal G. (1998), *Valuing the Future: Economic Theory and Sustainability*, Columbia University Press.

Hein L., van Koppen K., de Groot R. S. et van Ierland E. C. (2006), « Spatial scales, stakeholders and the valuation of ecosystem services », *Ecological Economics 57*:209-228.

Henry C. (2006), « Decision-making under scientific, political and economic uncertainty », *Cahiers du Laboratoire d'économétrie de l'École polytechnique*, n° DDX-06-12, 14 p.

Henry C. (1974), « Investment decisions under uncertainty: The 'irreversibility effect' », *American Economic Review 64* (Dec.), 1006–1012.

Henry C. et Henry M. (2006), « Formalization and application of the precautionary principle ». *Cahiers du Laboratoire d'économétrie de l'École polytechnique*, n° 2002-008, 19 p.

Jacobsen J. B. et Hanley N. (2009), « Are there income effects on global willingness to pay for biodiversity conservation? », *Environmental and Resource Economics* (in press) DOI 10.1007/s10640-008-9226-8.

Kahneman D. et Knetsch J. L. (1992), « Valuing public goods: The purchase of moral satisfaction », *Journal of Environmental Economics and Management 22* (1), 57-70.

Kahneman D. et Tversky A. (1982), « The psychology of preferences », *Scientific American 246*, 160-173.

Kassar I. et Lasserre P. (2004), « Species preservation and biodiversity value: A real option approach », *Journal of Environmental Economics and Management 48*, 857-879.

Kinzig A. P., Ryan P., Etienne M., Allison H., Elmqvist T. et Walker B. H. (2006), « Resilience and regime shifts: assessing cascading effects », *Ecology and Society 11*(1): 20.

Kinzig A., Perrings C. et Scholes B. (2007), « Ecosystem services and the economics of biodiversity conservation », *Working Paper*, School of Life Sciences, Arizona State University, 21 p.

Kirchhoff S., Colby B. G. et LaFrance J. T. (1996), « Evaluating the performance of benefit transfer: An empirical inquiry », *Journal of Environmental Economics and Management 33*, 75-93.

Kontoleon A., Pascual U. et Swanson T. (eds) (2007), *Biodiversity Economics: Principles, Methods and Applications*, Cambridge University Press, 664 p.

Kotchen M. J. et Reiling S. D. (2000), « Environmental attitudes, motivations, and contingent valuation of nonuse values: a case study involving endangered species », *Ecological Economics*, 32 (1), 93-107.

Krutilla J. V. (1967), « Conservation reconsidered », *American Economic Review*, 57 (4), 777-786.

Krutilla J. V. et Fisher A. C. (1975), *The Economics of Natural Environments: Studies in the Valuation of Commodity and Amenity Resources*, John Hopkins University Press, Baltimore, MD.

Lancaster K. (1966), « A new approach to consumer theory », *Journal of Political Economy*, 74, 132-157.

Larrère C. et Larrère R. (1998), *Du bon usage de la nature*, (*Discussing the Proper Use of Nature*) Aubier (« Alto »), Paris, 355 p.

Le Roux X., Barbault R., Baudry J., Burel F., Doussan I., Garnier E., Herzog F., Lavorel S., Lifran R., Roger-Estrade J., Sarthou J. P. et Trommetter M. (2008), *Agriculture et biodiversité. Valoriser les synergies* (Agriculture and Biodiversity. Valuing Synergies*)*, Expertise scientifique collective, synthèse du rapport, INRA, 90 p.

Leroux A. D., Martin V. L. et Goeschl T. (2008), « Optimal conservation, extinction debt, and the augmented quasi-option value », *10$^{th}$ BioEcon Worshop*, Cambridge (UK), September 29-30, 39 p.

Lifran R. et Oueslati W. (2007), « Éléments d'économie du paysage » (Elements of Landscape Economics), *Économie rurale (Rural Economics)*, n° 297-298, janvier-mars 2007.

Limburg K. E., O'Neill R. V., Costanza R. et Farber S. (2002), « Complex systems and valuation », *Ecological Economics*, 41 (3), 409–420.

Lockwood M. (1996), « Non-compensatory preference structures in non-market valuation of natural area policy », *Australian Journal of Agricultural Economics*, 40 (2), 85-101.

Loomis J. (1992), « The evolution of a more rigorous approach to benefit transfer: Benefit function transfer », *Water Resource Research*, 28, 701-705.

Loomis J. B. et White D. S. (1996), « Economic benefits of rare and endangered species: summary and meta-analysis », *Ecological Economics*, 18 (3), 197-/206.

Loomis J. B. et Rosenberger R. S. (2006), « Reducing barriers in future benefit transfers: Needed improvements in primary study design and reporting », *Ecological Economics*, 60 (2), 343-350.

Loomis J. B. *et al.* (2000), « Measuring the total economic value of restoring ecosystem services in an impaired river basin: Results from a contingent valuation survey », *Ecological Economics*, 33 (1), 103-117.

Loreau M., Mouquet N. et Gonzalez A. (2003), « Biodiversity as spatial insurance in heterogeneous landscape », *PNAS*, 100 (22), October 28, 12765-12770.

Loreau N., Naeem S., Inchausti P., Bengtsson J., Grime J. P., Hector A., Hooper D. U., Huston M. A., Raffaelli D., Schmid B., Tilman D. et Wardle D. A. (2001), « Biodiversity and ecosystem functioning: current knowledge and future challenges », *Science*, 294, 804-808.

Loreau N., Naeem S. et Inchausti P. (2002), *Biodiversity and Ecosystem Functioning: Synthesis and Perspectives*, Oxford, Oxford University Press.

Loureiro M. L. et Ojea E. (2008), « Valuing local endangered species: The role of intraspecies substitutes », *Ecological Economics*, 68 (2), 362-369.

Mahan B. L., Polasky S. et Adam R. M. (2000), « Valuing urban wetlands: A property price approach », *Land Economics*, 76 (1), 153-167.

Maille P. et Mendelsohn R. (1993), « Valuing Ecotourism in Madagascar », *Journal of Environmental Management*, 38, 213-218.

Mäler K.-G. (2008), « Sustainable development and resilience in ecosystems », *Environmental and Resource Economics*, 39, 17–24.

Mäler K.-G., Aniyar S. et Jansson Å (2009), « Accounting for ecosystems », *Environmental and Resource Economics*, 42 (1), 39–51.

Maris V. (2008), *La protection de la biodiversité: entre science, éthique et politique*, thèse pour le doctorat de philosophie, université de Montréal, Québec, 336 p.

Martens P., Rotmans J. et de Groot D. (2003), « Biodiversity: luxury or necessity? », *Global Environmental Change*, 13, 75-81.

Martinez-Alier J. (1995), « The environment as a luxury good or "too poor to be green"», *Ecological Economics*, 13 (1), 1-10.

Martinez-Alier J., Munda G. et O'Neill J. (1998), « Weak comparability of values as a foundation for ecological economics », *Ecological Economics*, 26, 277–286.

McCann K. S. (2000), « The diversity-stability debate », *Nature*, 405, 228-233.

McComb G., Lanz V., Nash K. et Rittmaster R. (2006), « International valuation databases: Overview, method and operational issues », *Ecological Economics*, 60 (2), 461-472.

McConnell K. E. (1997), « Does altruism undermine existence value? », *Journal of Environmental Economics and Management*, 32, 22-37.

McNeely J. A. (1988), *Economics and Biological Diversity: Developing and Using Economic Incentives to Conserve Biological Resources*, IUCN, Gland, Switzerland.

Merlo M. et Croitoru L. (2005), *Valuing Mediterranean Forests: Toward Total Economic Value*, Cabi Publishing, Wallingford Oxfordshire, UK.

Meier C. E. et Randall A. (1991), « Use value under uncertainty: Is there a « correct » measure? », *Land Economics*, 67 (4), 379-389.

Metrick A. et Weitzman M. L. (1998), « Conflicts and choices in biodiversity Preservation », *Journal of Economic Perspective*, 12 (3), Summer, 21-34.

Millennium Ecosystem Assessment (2005a), *Ecosystems and Human Well-being: A Framework for Assessment*, Island Press, 201 p.

Millennium Ecosystem Assessment (2005b), *Ecosystems and Human Well-Being: Synthesis*, Island Press, 137 p.

Milon J. W. et Clemmons R. (1991), « Hunters' demand for species variety », *Land Economics*, 67 (4), 401-412.

Mitchell R. C. et Carson R. T. (1981), *An Experiment in Determining WTP for National Water Quality Improvements*, Resources for the Future, Inc., Washington D.C.

Moran A. (1992), « Scope and limits of the property-rights approach to species conservation », *Policy*, 8, 22–27.

Montagné C., Peyron J.-L. et Niedzwiedz A. (2005), *La valeur économique totale de la forêt méditerranéenne française*, (Overall Economc Value of the French Mediterranean Forest) MEDFOREX.

Moyle B. (1998), « Species conservation and the principal–agent problem », *Ecological Economics*, 26, 313–320.

Musgrave R. (1987), « Merit goods », *in* Eatwell J., Millgate M. et Neuman P. (ed.), *The New Palgrave: a Dictionnary of Economics*, MacMillan, London, 452-453.

Naidoo R., Balmford A., Costanza R., Fisher B., Green R. E., Lehner B., Malcolm T. R. et Ricketts T. H. (2008), « Global mapping of ecosystem services and conservation priorities », *PNAS*, 105 (28), 9495–9500.

Narain U., Hanemann M. et Fisher A. (2007), « The irreversibility effect in environmental decisionmaking », *Environmental and Resource Economics*, 38, 391-405.

Nehring K. et Puppe C. (2002), « A theory of diversity », *Econometrica*, 70 (3), 1155-1198.

Nelson J. P. et Kennedy P. E. (2009), « The use (and abuse) of meta-analysis in environmental and natural resource economics: An assessment », *Environmental and Resource Economics*, in press, DOI 10.1007/s10640-008-9253-5.

Neumayer E. (1999), *Weak versus Strong Sustainability: Exploring the Limits of Two Opposing Paradigms*, Edward Elgar, Cheltenham.

Nijkamp P., Vindigni G. et Nunes P. A. L. D. (2008), « Economic valuation of biodiversity: A comparative study », *Ecological Economics*, doi:10.1016/j.ecolecon.2008.03.003.

Norgaard R. B. (1988), « The rise of the global exchange economy and the loss of biological diversity », *in* Wilson E. O. (ed.), *Biodiversity*, The National Academies Press, Washington DC.

Norton B. (ed.) (1986), *The Preservation of Species: The Value of Biological Diversity*, Princeton, Princeton University Press.

Norton B. G. (1987), *Why Preserve Natural Variety?*, Princeton University Press, Princeton, NJ, 281 p.

Nunes P. (2002), « Using factor analysis to identify consumer preferences for the protection of a natural area in Portugal », *European Journal of Operational Research*, 140, 499-516.

Nunes P. et Schokkaert E. (2003), « Identifying the warm glow effect in contingent valuation », *Journal of Environmental Economics and Management*, 45 (2), March, 231-245.

Nunes P. A .L. D. et van den Bergh J. C. J. M. (2001), « Economic valuation of biodiversity: sense or nonsense ? », *Ecological Economics*, 39 (2), 203-222.

Nunes P. A. L. D., van den Bergh J. C. J. M. et Nijkamp P. (2003), *The Ecological Economics of Biodiversity. Methods and Policy Applications*, Edward Elgar Publishing, Cheltenham, 165 p.

OCDE (2002), *Manuel d'évaluation de la biodiversité – Guide à l'intention des décideurs, (Biodiversity Assessment Manual - A Guide for Decision Makers)* OCDE, Paris, 173 p.

OCDE (2001), *Valuation of Biodiversity Benefits: Selected Studies*, Paris, OCDE.

Odum W. E. (1982), « Environmental degradation and the tyranny of small decisions », *BioScience*, 32 (9); 728-729.

Ohl C., Drechsler M., Johst K. et Wätzold F. (2008), « Compensation payments for habitat heterogeneity: existence, efficiency, and fairness considerations, *Ecological Economics*, 67, 162-174.

Ojéa E. et Loureiro M. L. (2007), « Altruistic, egoistic and biospheric values in willingness to pay (WTP) for wildlife », *Ecological Economics*, *63* (4), September, 807-814.

O'Neill J., Holland A. et Light A. (2007), *Environmental Values*, Routledge, 233 p.

Opaluch J. J. et Grigalunas T. A. (1992), *Ethical Values and Personal Preferences as Determinants of Nonuse Values: Implications for Natural Resource Damage Assessments*, W133 Meetings Fifth Interim Report.

Ostrom E., Gardner R. et Walker J. (1994), *Rules, Games and Common-Pool Resources*, Ann Arbor, University of Michigan Press.

Page T. (1977), *Conservation and Economic Efficiency*, Baltimore, John Hopkins University Press.

Pagiola S., von Ritter K. *et al.* (2004), « Assessing the economic value of ecosystem conservation », World Bank, *Environment Department Paper No.101*, in collaboration with The Nature Conservancy and IUCN.

Parkhurst G., Shogren J F., Bastian C., Kivi P., Donner J. et Smith R. B. W. (2002), « Agglomeration bonus: an incentive mechanism to reunite fragmented habitat for biodiversity conservation », *Ecological Economics*, 41, 305-328.

Patterson M. (1998), « Commensuration and theories of value in ecological economics », *Ecological Economics*, 25, 105-125.

Pearce D. W. (2007), « Do we really care about biodiversity? », *Environmental and Resource Economics*, 37, 313-333.

Pearce D. W. (2001), « The economic value of forest ecosystems », *Ecosystem Health*, 7 (4), 284-296.

Pearce D. W. (1976), « The limits of cost-benefit analysis as a guide to environmental policy », Kyklos, fasc. 1. (repris *in* D.W. Pearce, *Economics and the Environment: Essays in Ecological Economics and Sustainable Development*, Edward Elgar, Cheltenham, 1999).

Pearce D. et Moran D. (1994), *The Economic Value of Biodiversity*, IUCN and Earthscan.

Perman R., Ma Y., McGilvray J. et Common M. (2003), *Natural Resource and Environmental Economics*, 3rd edition, Pearson Education Limited, Harlow, 699 p.

Perrings C. (2006), « Ecological economics after the Millennium Assessment », *International Journal of Ecological Economics & Statistics*, 6, No. F06, 8-22.

Perrings C. A. (1998), « Resilience in the dynamics of economy-environment systems », *Environmental and Resource Economics*, 11 (3-4): 503-520.

Perrings C. (1995a), « Economic values of biodiversity », *in* Heywood V. H., Watson R. T. *et al.*, *Global Biodiversity Assessment*, UNEP and Cambridge University Press, Cambridge (UK), 823-914.

Perrings C. (1995b), « Biodiversity conservation as insurance », *in* Swanson T. (ed.), *The Economics and Ecology of Biodiversity Decline*, Cambridge, Cambridge University Press, 69-78.

Perrings C. et Gadgil M. (2003), « Conserving biodiversity: reconciling local and global public benefits », *in* Kaul I. (ed.), *Providing Global Public Good: Making Globalization Work for All,* Oxford University Press, Oxford, UK.

Perrings C., Mooney H. et Williamson M. (eds) (2009), *Bioinvasions and Globalization: Ecology, Economics, Management, and Policy*, Oxford University Press, Oxford (in press).

Perrings C., Mäler K.-G., Folke C., Holling C. S. et Jansson B. O. (1995), *Biodiversity Loss. Economic and Ecological Issues*, Cambridge University Press, Cambridge and New York.

Perrings C. et Pearce D. (1994), « Threshold effects and incentives for the conservation of biodiversity », *Environmental and Resource Economics,* 4, 13-28.

Perrings C., Williamson M. H. et Dalmazzone S. (2000), *The Economics of Biological Invasions*, Edward Elgar Publishing, Heltenham, UK.

Pimentel D. (ed.) (2002), *Biological Invasions. Economic and Environmental Costs of Alien Plant, Animal, and Microbe Species*, CRC Éditions, 370 p.

Pimentel D., Wilson C., McCullum C., Huang R., Dwen P., Flack J., Tran Q., Saltman T. et Cliff B. (1997), « Economic and environmental benefits of biodiversity », *BioScience*, 47, 747-757.

Plott C. R. (1996), « Rational individual behavior in markets and social choice processes: The discovered preference hypothesis », *in* Arrow K., Colombatto E., Perlaman M. et Schmidt C. (eds), *The rationality foundations of economic behavior*, Macmillan and St. Martin's Press, London and New York, 225–250.

Point P. (ed.) (1999), *La valeur économique des hydrosystèmes. Méthodes et modèles d'évaluation des services délivrés*, ( *Economic Value of Ecosystems. Methods and Models for Assessing the Provided Services)* Paris, Economica, 211 p.

Polasky S. (ed.) (2002), *The Economics of Biodiversity Conservation,* Ashgate, London.

Polasky S., Camm J. D. et Garber-Yonts B. (2001), « Selecting biological reserves cost-effectively: an application to terestrial vertabrate conservation in Oregon »; *Land Economics*, 77 (1), 68-78.

Polasky S., Camm J., Solow A., Csuti B., White D. et Ding R. (2000), « Choosing reserve networks with incomplete species information », *Biological Conservation*, 94, 1-10.

Polasky S., Costello C. et Solow A. (2005), « The economics of biodiversity conservation », *in* Vincent J. et Mäler K.-G. (eds), *The Handbook of Environmental Economics*, North-Holland, 1517-1560.

Polasky S. et Solow A. (1995), « On the value of a collection of species », *Journal of Environmental Economics and Management*, 29, 298-303.

Polasky S. et Solow A. (1999), « Conserving biological diversity with scarce resources », *in* Klopatek J. et Gardner R. (ed.), *Landscape Ecological Analysis: Issues and Applications*, Springer Verlag, New York, 154-174.

Portney P. R. et Weyant J. P. (eds) (1999), *Discounting and Intergenerational Equity*, Resources for the Future, Washington DC, 186 p.

Purvis A. et Hector A. (2000), « Getting the measure of biodiversity », *Nature*, 405, 212-219.

Rambonilaza M. (2004), « Évaluation de la demande de paysage: état de l'art et réflexions sur la méthode du transfert des bénéfices »(Assessment of the Landscape Demand: State of the Art and Reflections on the Profit Transfer Method), *Cahiers d'économie et sociologie rurales*, n° 70, p. 77-101.

Randall A. (1991), « The value of biodiversity », *Ambio*, 20 (2), 64-68.

Randall A. et Farmer M. C. (1995), « Benefits, costs, and the safe minimum standard of conservation », *in* Bromley D. W. (ed.), *The Handbook of Environmental Economics*, Oxford UK and Cambridge États-Unis, Blackwell Publishers, 26-44.

Rausser G. C. et Small A. A. (2000), « Valuing research leads: bioprospecting and the conservation of genetic resources », *Journal of Political Economy*, 108 (1): 173-206.

Rawls J. (1971), *A Theory of Justice*, Oxford University Press, Cambridge, MA.

Ready R. C. et Bishop R. C. (1991), « Endangered species and the safe minimum standards », *American Journal of Agricultural Economicsi*, 73 (2), 309-312.

Redclift M. (1993), « Sustainable development: needs, values, rights », *Environmental Values*, 2(1), 3-20.

Rees W. E. (1992), « Ecological footprints and appropriated carrying capacity: what urban economics leaves out », *Environment and Urbanisation*, 4 (2), 121–130.

Rekola M. (2003), « Lexicographic preferences in contingent valuation: A theoretical framework with illustration », *Land Economics*, 79 (2), 277-291.

Richardson L. et Loomis J. (2009), « The total economic value of threatened, endangered and rare species: An updated meta-analysis », *Ecological Economics*, doi:10.1016/j.ecolecon.2008.10.016.

Ricketts T. H., Daily G. C., Ehrlich P. R. et Michener C. D. (2004), « Economic value of tropical forest to coffee production », *PNAS*, 101 (34), 12579–12582.

Riera P. (2001), *Assessment of Methodologies for Valuing Biological Diversity of Forests*, European Forest Institute, Internal Report No. 4.

Riera P. et Mogas J. (2004), « Finding the social value of forests through stated preference methods. A Mediterranean forest valuation exercise », *Silva Lusitana*, N° especial, 17-34.

Rosenberger R. S., Peterson G., Clarke A. et Brown T. (2003), « Measuring disposition for lexicographic preferences of environmental goods: integrating economics, psychology and ethics », *Ecological Economics*, 44 (1), 63-76.

Rozan A. et Stenger A. (2000), « Intérêts et limites de la méthode du transfert de bénéfices », ("Interest and Limits of the Benefits Transfer Method") *Économie et Statistique)*, n° 336, p. 69-78.

Rulleau B. (2008), *Services récréatifs en milieu naturel littoral et évaluation économique multi-attributs de la demande (Recreational Services in a Natural Shore Environment and Multi-Attribute Economic Assessment of Demand)*, thèse de doctorat es sciences économiques, université Montesquieu -Bordeaux IV.

Sagoff M. (2008), « On the economic value of ecosystem services », *Environmental Values 17*, 239–257.

Sagoff M. (2004), *Price, Principle and the Environment*, Cambridge University Press, Cambridge (UK).

Sagoff M. (1998), « Aggregation and deliberation in valuing environmental public goods: A look beyond contingent pricing », *Ecological Economics*, 24 (2-3): 213-230.

Sandler T. (1998), « Global and regional public goods: a prognosis for collective action », *Fiscal Studies*, 19, 221–247.

Sandler T. et Arce D. G. (2003), « Pure public good versus commons: benefit-cost duality », *Land Economics*, 79 (3), 355-368.

Sarr M., Goeschl T. et Swanson T. (2008), « The value of conserving genetic resource for R&D: A survey », *Ecological Economics*, 67, 184-193.

Schläpfer F. (2008), « Contingent valuation: A new perspective », *Ecological Economics*, 64, 729-740.

Schläpfer F., Roschewitz A. et Hanley N. (2004), « Validation of stated preferences for public goods: a comparison of contingent valuation survey response and voting behaviour », *Ecological Economics*, 51 (1-2), 1-16.

Sen A. (1995), « Environmental evaluation and social choice: contingent valuation and the market analogy », *Japanese Economic Review*, 23 (1), 23-37.

Sen A. K. (1987), *On Ethics and Economics*, Blackwell, Oxford.

Sen A. K. (1977), « Rational fools: A critique of the behavioural foundations of economic theory », *Philosophy and Public Affairs*, 16, 317-344.

Senbel M., McDaniels T. et Dowlatabadi H. (2003), « The ecological footprint: a non-monetary metric of human consumption applied to North America », *Global Environmental Change*, 13. 83–100.

Serre M. (1987), *Le Contrat naturel*, François Bourin, Paris, 191 p.

Shogren J .F., Shin S. Y., Hayes D.J. et Kliebenstein J. B. (1994), « Resolving differences in willingness to pay and willingness to accept », *American Economic Review*, 84 (1), 255-270.

Shogren J. F. et Tschirhart J. (2005), « Integrating ecology and economics to address bioinvasions », *Ecological Economics*, 52 (3) 267–271.

Simpson R. D. (2007), « David Pearce and the economic valuation of biodiversity », *Environmental and Resource Economics*, 37 (1), 91–109.

Simpson R. D., Sedjo R. et Reid J. W. (1996), « Valuing biodiversity for use in pharmaceutical research », *Journal of Political Economy*, 104 (1), February, 163-185.

Smith V. K. (1992), « Arbitrary values, good causes, and premature verdict », *Journal of Environmental Economics and Management*, 22, 71-89.

Smith V. K. et Kaoru Y. (1990), « Signals or noise? Explaining the variation in recreation benefit estimates », *American Journal of Agricultural Economics*, 72 (2), 419-433.

Smith V. K. et Osborne L. L. (1996), « Do contingent valuation pass a “scope” test? A meta-analysis », *Journal of Environmental Economics and Management*, 31, 287-301.

Smith V. K., Pattanayak S. K. et van Hotven G. L. (2006), « Structural benefit transfer: An example using VSL estimates », *Ecological Economics*, 60 (2), 361-371.

Solow A. R., Polasky S. et Broadus J. M. (1993a), « On the measurement of biological diversity », *Journal of Environmental Economics and Management*, 24, 60-68.

Solow A. R., Polasky S. et Broadus J. M. (1993b), « Searching of uncertain benefits and the conservation of biological diversity », *Environmental and Resource Economics*, 3, 171-181.

Spash C. L. et Hanley N. D. (1995), « Preferences, information and biodiversity preservation », *Ecological Economics*, 12, 191-208.

Spash C. L. et Vatn A. (2006), « Transferring environmental value estimates: Issues and alternatives », *Ecological Economics*, 60 (2), December, 379-388.

Spash C., Urama K., Burton R., Kenyon W., Shannon P. et Hill G. (2009), « Motives behind willingness to pay for improving biodiversity in a water ecosystem: Economics, ethics and social psychology », *Ecological Economics*, *68* (4), 955-964.

Stenger A., Harou P. et Navrud S. (2009), « Valuing environmental goods and services derived from the forests », *Journal of Forest Economics*, 15, Special Issue: Valuing Environmental Goods and Services Derived from the Forests, 1–14.

Sukhdev P. (ed.) (2008), *The Economic of Ecosysems and Biodiversity*, Interrim Report, European Community, 68 p.

Stevens T. H., Etchevarria J., Glass R.J., Hager T. et More T. A. (1991), « Measuring the existence value of wildlife: what do CVM estimates really show? », *Land Economics*, 67, 390-400.

Swanson T. M. (ed.) (1995), *The Economics and Ecology of Biodiversity Decline. The Forces Driving Global Change*, Cambridge University Press, Cambridge and New York.

Swanwick C., Hanley N. et Termansen M. (2007), *Scoping Study on Agricultural Landscape Valuation*, Final report to DEFRA.

Tacchoni L. (2000), *Biodiversity and Ecological Economics. Participation, Values and Resource Management*, Earthscan, London and Sterling, VA.

Thomas A., Gozlan E. et Loope L. (2006), « Impacts de l’introduction d’espèces envahissantes en Nouvelle-Calédonie: aspects méthodologiques pour l’évaluation économique et pistes de réflexion pour un partage efficace des coûts » (Impacts of Introducing Invasive Species into the New Caledonia Archipelago), *in* Beauvais M.-L., Coléno A. et Jourdan H. (éds), *Les espèces envahissantes dans l’archipel néo-calédonien.*, Éditions IRD (« Expertise collégiale »), Paris.

Thoyer S. et Said S. (2007), « Mesures agri-environnementales: quels mécanismes d’allocation ? (Agri-Environmental Measures: which Allocation Mechanisms Should Be Used ?)», *in* Doussan I. et Dubois J. (eds), *Conservation de la biodiversité et Politique agricole commune de l’Union européenne*, Paris, La Documentation française.

Tilman D. (1999), « The ecological consequences of changes in biodiversity: A search for general principles », *Ecology*, 80 (5), 1455-1474.

Tilman D., May R. M., Lehman C. L. et Nowak M. A. (1994), « Habitat destruction and extinction debt », *Nature*, 371, 65-66.

Tilman D., Polasky S. et Lehman C. (2005), « Diversity, productivity and temporal stability in the economies of human and nature », *Journal of Environmental Economics and Management*, 49 (3), 405-426.

Toman M. (1998), « Why not to calculate the value of the World's ecosystem services and natural capital », *Ecological Economics*, 25 (1), 57-60.

Trommetter M. (2005), « Biodiversity and international stakes: a question of access? », *Ecological Economics*, 53, 573-583.

Turner R. K. et Daily G. C. (2008), « The ecosystem services framework and natural capital conservation », *Environmental and Resource Economics*, 39, 25-35.

Turner R. K., Paavola J., Cooper P., Farber S., Jessamy V. et Georgiou S. (2003), « Valuing nature: lessons learned and future research directions », *Ecological Economic*, 46, 493-510.

Turpie J. K. (2003), « The existence value of biodiversity in South Africa: How interest, experience, knowledge, income and perceived level of threat influence local willingness to pay », *Ecological-Economics*, 46 (2), 199-216.

Tversky A. et Kahneman D. (1981), « The framing of decision and the psychology of choice », *Science*, 211, 453-458.

Van Kooten G. C. (1998), « Economics of conservation biology: a critical overview », *Environmental Science and Policy*, 1, 13-25.

Van Kooten G. C. et Bulte E. H. (2000a), *The Economics of Nature: Managing Biological Assets*, Oxford, U, Blackwell Publishers, 512 p.

Van Kooten G. C. et Bulte E. H. (2000b), « The ecological footprint: useful science or politics », *Ecological Economics 32* (3), 385-389.

Vatn A. et Bromley D. W. (1995), « Choices without prices without apologies », *in* Bromley D. W. (ed.), *The Handbook of Environmental Economics*, Oxford (UK) and Cambridge (USA), Blackwell Publishers, 3-25.

Vitouzek P., Mooney H., Lubchenko J. et Melillo J. (1997), « Human domination of the Earth ecosystems », *Science*, 277, 494-499.

Wackernagel M., Onisto L., Bello P., Linares A. C., Falfán I. S. L., García J. M., Guerrero A. I. S., et Guerrero M. G. S. (1999), « National natural capital accounting with the ecological footprint concept », *Ecological Economics*, 29 (3), 375-390.

Walker B., Holling C.S., Carpenter S. R. et Kinzig A. (2004), « Resilience, adaptability and transformability in social–ecological systems », *Ecology and Society*, 9 (2), 5.

Walker B. H., Gunderson L.H., Kinzig A. P., Folke C., Carpenter S.R. et Schultz L. (2006), « A handful of heuristics and some propositions for understanding resilience in social-ecological systems », *Ecology and Society*, 11 (1), 13.

Wallace K. J. (2007), « Classification of ecosystem services: problems and solutions », *Biological Conservation*, 139 (3–4), 235–246.

Walsh R. O., Loomis J. B. et Gillman R. A. (1984), « Valuing option, existence, and bequest demands for wilderness », *Land Economics*, 60, 14–29.

Walsh R., Bjonback R. *et al.* (1990), « Estimating the public benefits of protecting forest quality », *Journal of Environmental Management*, 30 (2), 175-189.

Weber J. (2003), « Évaluation contingente: les valeurs ont-elles un prix ? »(Contingent Evaluation: Can Value Be Priced ?), *Compte-rendus de l'Académie d'Agriculture*, février 2003.

Weikard H.-P. (2002), « Diversity functions and the value of biodiversity », *Land Economics*, 78 (1), 20-27.

Weisbrod B. A. (1964), « Collective-consumption service of individual-consumption goods », *Quarterly Journal of Economics*, 77, 471-477.

Weitzman M. L. (2000), « Economic profit versus ecological entropy », *Quarterly Journal of Economics*, 115 (1), 237-263.

Weitzman M. L. (1998), « The Noah's Ark problem », *Econometrica*, 66 (6), November, 1279-1298.

Weitzman M. L. (1993), « What to preserve? An application of the diversity theory to crane conservation », *Quarterly Journal of Economics*, 108 (1), February, 157-183.

Weitzman M. L. (1992), « On diversity », *Quarterly Journal of Economics*, 107 (2), 363-405.

Westerberg V. et Lifran R. (2008), « Eliciting social preferences and trade-off in landscape using choice experiment: Pilot study in the anciens marais des Beaux, Provence, France », *10$^{th}$ BioEcon Worshop*, Cambridge (UK), September 29-30, 34 p.

Westman W. (1977), « How much are nature's services worth », *Science*, 197, 960–964.

Willig R. D. (1976), « Consumer surplus without apology », *American Economic Review*, 66, 589-597.

Willinger M. (1996), « La méthode d'évaluation contingente: de l'observation à la construction des valeurs de préservation »(, (Contingent Evaluation Method: from Observation to the Construction of Preservation Values) *Nature, Science et Société*, janvier.

Willis K. G. et Garrod G. (1993), « Valuing landscape: a contingent valuation approach », *Journal of Environmental Management*, 37, 1-22.

Willis K. G., Garrod G. et Saunders C. M. (1995), « Benefits of environmentally sensitive area policy in England: a contingent valuation assessment », *Journal of Environmental Management*, 44, 105-125.

Willis K. G., Garrod G., Scarpa R., Powe N., Lovett A., Bateman I J., Nanley N. et Macmillan D. C. (2003), *The Social and Environmental Benefits of Forests in Great Britain*, Report CREAM to Forestry Commission, Newcastle, July, 36 p.

Wilson M. A. et Howarth R. B. (2002), « Discourse-based valuation of ecosystem services: establishing fair outcomes through group deliberation », *Ecological Economics*, 41, 431-443.

Wilson M. A. et Hoehn J. P. (2006), « Valuing environmental goods and services using benefit transfer: The state-of-the art and science », *Ecological Economics*, 60 (2), December, 335-342.

Woodward R. T. et Wui Y. S. (2001), « The economic value of wetland services: A meta-analysis », *Ecological Economics*, 37, 257-270.

Wu J. et Bogess W. G. (1999), « The optimal allocation of conservation funds », *Journal of Environmental Economics and Management*, 38, 302-321.

Xue D. et Tisdell C. (2001), « Valuing ecological functions of biodiversity in Changbaishan Mountain biospere reserve in northeast China », *Biodiversity and Conservation*, 10, 467-481.

Zandersen M. et Tol R. S. J. (2009), « A meta-analysis of forest recreation values in Europe », *Journal of Forest Economics*, 15, 109-130.

Zhang W., Ricketts T.H., Kremen C., Carney K. et Swinton S. M. (2007), « Ecosystem services and dis-services to agriculture », *Ecological Economics*, 64, 253-260.

# Chapter VI

# Summary of research needs

In the past few decades, research on biodiversity has accelerated substantially with the perception of rising stakes and the emergence of new research tools, both to understand its functioning and to analyse the threats that it is facing (and the means to oppose them).

This chapter will limit itself to identifying the unsatisfied needs in terms of understanding of the socio-economic stakes linked to biodiversity and the taking into account of the services of ecosystems in the socio-economic evaluation of projects and/or public policies. The monetarisation of biodiversity and its use in political decision-making processes raise various questions regarding research and also both theoretical and empirical controversies, which were already largely detailed in the chapters on the state of knowledge. The working group deliberately undertook an approach based on a substantial international academic bibliography, in accordance with the state of the art and current mobilisations.

**Understanding the mechanisms that explain the dynamics of biodiversity, including its dynamics in anthropised milieus, remains a priority for which the modalities are presented in the National Research Strategy on Biodiversity.** Moreover, it is essential to deepen our understanding of the mechanisms of adaptation to climate change and the attenuation of its effects, which are both a factor of interference with the reference system (what is natural or man's responsibility?) and a challenge (do the phenomena at work represent, even independently of the action of man, a loss of "capital" that it would be legitimate to fight?). The research aspects of domestic and European strategies concerning biodiversity focus mostly on these aspects and will provide knowledge both in terms of mechanisms and inventory.

Moreover, analysis of the contributions of biodiversity and ecosystems to the creation of value and to economic and social life is currently seeing substantial developments on a global level. A large number of actions, reports, and studies were analysed and included with the references of the working group. This work is more or less sound and rigorous in terms of economic theory. **Certain major initiatives have as their major efficacy, for the time being, the promoting of collective consciousness-raising regarding the stakes. They only rarely supply tools for the integration of values in public decision-making processes**.

Given the current state of the work, these complexities and the unknowns on several dimensions of biodiversity were obstacles to the progress of the working group. They are thus part of the pending research needs.

The text of the mission does not ignore this situation, because it requests both an overview of knowledge and research proposals. It thus frames the request in an outlook that is as operational as possible. These suggestions do not claim to represent an inventory of the general research needs for economics, ecology, nor even for the value of biodiversity[1]. The idea is simply to formulate proposals to make the mobilisation of economic concepts in the protection and management of biodiversity in France as operational as possible, particularly in the technical-administrative and political reasoning implemented for the choice of infrastructure.

Thinking on the stakes for society and the definition and analysis of possible policies to be implemented also imply sophisticated skills in social sciences. **Knowledge of the socio-economic stakes of biodiversity and normative analysis of the desirable objectives and means to achieve them remain very fragmented.** On this point, we recall the recommendations from the meeting of the European Platform for a research strategy in biodiversity (EPBRS, recommendations in appendix).

Research oriented towards the operational integration of the value of biodiversity in public decision-making processes, or even private ones, could usefully involve thematic segmentations for monitoring of the financing of research proposed by the Operational Research Committee stemming from the *Grenelle de l'environnement*.

The proposals cover two major areas:

- The constitution of data, information, follow-up;
- The development and enhanced reliability of knowledge (methods, socio-economic reference systems) in social sciences.

We should point out that the values that can be calculated by various complex and heterogeneous economic methods do not encompass all of the values of all orders that are involved in the final decision. **The extension of the exercise of monetarisation to the values that are most mobilising for the preservation of biodiversity (legacy, functional role, scarcity) by rejecting their incommensurability is a highly debatable process, and the report details its current limitations.**

**In any case, and in general, the fragility of current knowledge of ecological functioning and the interactions with human activities constitute a major limitation. This fragility is behind many of the orientations of the national strategy for biodiversity.**

## 1. Reinforcing links with international programmes

International expertise, inter-governmental mobilisation, objectification of knowledge and outlooks, presentation of results in terms of certainties, probabilities and risks: these are the elements that allowed climate change to become part of decision-makers' concerns. The GIEC system was determining. The issue of biodiversity presents similar challenges. The process of consultation towards an IMoSEB (*International Mechanism of Scientific Expertise on Biodiversity*) launched in 2005 identified the needs and options of such a system for biodiversity, from the international to the more local levels (see www.imoseb.net). The coming together of

[1] For this, see the work of the Scientific Committee of the IFB (2008), *Strategic Thinking – Report of thinking groups*, IFB Paris, 80 p. (available for downloading on the site www.fondationbiodiversite.fr/Accueil.html).

this initiative supported at the outset by France and the continuation of the MEA is currently borne by the PNUE under the term IPBES (*Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services*). The need to reinforce the interface between knowledge, opinion and decisions is unanimously acknowledged, even in diplomatic speeches. The means to achieve this on an international level remain to be negotiated. The progressive integration of biodiversity on all economic and political levels constitutes the central factor. The very nature of many of the stakes of biodiversity implies an international vision, as most of the actors are international (NGO's, scientific organisations, jurisdictions).

Research on biodiversity must, like the issue of climate change, remain mobilised in the international systems of collective expertise, with the objective of appropriation by public authorities of the observations on the evolution of biodiversity in the world and on each body's potential for action. Lastly, institutional evolutions will be necessary to build interfaces and French scientific capacity in these fields: the statuses and general orientations of the research establishments (INRA, CEMAGREF, IFREMER, MNHN, etc.) must better specify the place of biodiversity in their missions.

## 2. Reinforcing the production of data for the determination of reference values

**We must distinguish two orientations: first, the stakes linked to methodological developments, which familiarise the actors with the services of ecosystems and promote the elaboration of preferences through evaluation procedures allowing for exchanges between actors and experts; also, the challenge presented by the lack of concrete work applying the available methods to spaces in the country that could reinforce the skills of experts.** This point is important because it means that academic dynamics, and the incentive systems that depend on them, may not be able, or at least not on their own – to make up for this lack. We note in particular the weakness – in comparison with tropical milieus – of studies devoted to the services of ecosystems of temperate zones and the disproportion between certain highly documented services, such as the fixing of carbon or recreational value, and other studies for which estimations are rare, such as protection functions. In this latter case, coupled approaches, involving specialists of the physical milieu, ecology, economics and risk management should probably be promoted. The issue arises for nature risks but also for human health, in its link with biodiversity and the environment, whether for modulation of the presence or the effect of pathogenic agents or polluting substances.

### 2.1. Reinforcing French references and drawing the interest of French administrative and scientific leaders in these fields

**The meta-analyses now available include few values established on French contexts (Metropolitan and Overseas)**, although France has recognised specialists and emblematic situations (substantial coastline, many biogeographic combinations, intermediary territory in terms of population density, diversified Overseas areas). This is partly due to a lack of training in economic sciences and a problem of recognition of this type of work in the French scientific community. For planning studies, it is desirable that public sponsors be better trained in the economic evaluation of biodiversity: this involves the various places for the training of civil servants (IFORE, PNF, PNR, Network of the major sites of France). It is firstly necessary that studies of direct impacts (such as those of RGSF) and the economic or socio-economic evaluations of biodiversity be done more often and above all within the entities that should be the depositories of the latest position papers such as the PNR, RNF, RGSF, CELR. Also, appropriate training programmes must be developed so that the sponsors (new DREAL and DDEA, for example) can better grasp the value of these methods, order them and use them. These last two recommendations are not strictly matters of research, but obviously introduce issues of the means of integrating biodiversity in the action of the State – issues that remind us that nothing is obvious in this field in terms of knowledge, aims, methods and means of action.

**Geographic priorities must be set: Mediterranean biogeographic zone, Overseas Departments, etc.**

The report of the Sénat, *The contributions of science and technology to sustainable development, volume II: Biodiversity: the other shock? The other chance?*, in its section "One of the tool boxes of the fourth industrial revolution", reminds us that the "memory of success" that the biodiversity of living systems constitutes should lead to industrial development based on biology and biotechnology. It points out at the same time that France shows **too little interest in the biological mechanisms that could be one of the driving forces of the next industrial revolution.**

## 2.2. Developing French capacity for mobilisation for the purpose of evaluation of the data from the knowledge and observation of ecological phenomena

Multiple projects involving knowledge of biodiversity (inventory, mechanisms of evolution) are producing a substantial quantity of data. The economic evaluation can mobilise them in order to produce reference values, through meta-analyses, as long as the conditions for obtaining and publishing the values allow for this mobilisation.

**Dependency on local conditions and on the methodological conception of the evaluations of ecosystems and biodiversity is an obstacle to the generalisation of the values obtained** in particular cases – which we call the issue of the transfer of value. The generalisation of the use of an economic value of biodiversity in socio-economic calculations prior to making public decisions assumes a certain exhaustiveness in the types of elements of biodiversity monetarised in order to cover (either through the transfer of reference values, or by the possibility of developing indicators of spatial modulation of these values):

- The largest territory;
- The most diverse situations and ecosystems.

The research needs will involve in particular the multiplication of case studies in conditions for the development of data to enrich databases such as the "Environmental Valuation Reference Inventory" (EVRI) (www.evri.ca/), developed by Environment Canada and supported by MEEDAT in particular, or the "Case Study Database", developed by the "Nature Valuation Network" (www.fsd.nl/naturevaluation/73766).

The idea is to address two obstacles to the generalisation of the partial values reachable with the current state of knowledge:

- Extending the known services and the conditions for their evolution in cases of public decision-making, mobilisation of economic tools, realisation of infrastructure and human pressure;
- Correcting the difficulties of geographic transposability and changes of scale in the mobilisation of values (this issue remains as determining for setting the limits and validating the conditions of relevance for reparation, substitution, and compensation actions).

This is mainly the area covered by the Millennium Ecosystem Assessment and by all of the studies concerning the functionalities and productivity of the ecosystems. The French proposals for the MEA should thus be supported. Beyond observation, the understanding and analysis of fields must be enhanced and particularly the functioning of soils (agricultural land, forests, polluted soils, etc.) and of aquatic systems (priority displayed by Grenelle).

**Nevertheless, restriction to only the values of ecosystemic services or to a purely utilitarian rationale would present the insufficiencies that are frequently cited** (see the appendix concerning the recommendations from the meeting of the European Platform for a biodiversity research strategy, EPBRS): **means to determine the intrinsic values of biodiversity, etc., must be developed.**

## 2.3. Very broadly incorporating data from socio-economic observations, indicators of ecological services and impacts of human activities in long-term observation systems

Observation thus remains a determining element of the capacity to provide expertise to support public policies to:

- Document the evolution of the loss of biodiversity (quantified data, indicators, scenarios, maps, collections, etc.);
- Anticipate the consequences of the loss of biodiversity by assembling information concerning both the state of biodiversity and the impacts on society;
- Provide data to monitor the evaluation over time.

The establishment of the ONEMA (Office national de l'eau et des milieux aquatiques – National office of water and aquatic milieus), a public establishment which has as one of its central missions the construction of a national database on water and aquatic milieus, constitutes an example of a recent initiative in this direction.

The ORE (Observatoires régionaux de l'environnement – Regional Environment Observatories), developed in 2001 and established in 2003 in France, are long-term observatories formed on the model of the American environmental observatories (LTER): they are among the systems that can progressively offer the means to identify the dynamics at work on biodiversity, like the European LTER. In order to allow for comparisons of the evolution of ecosystems and socio-ecosystems over time, it is essential to implement systems for fine and repeated measurements on various types of ecosystems/socio-ecosystems representative of the various biogeographic and geoeconomic situations. On the international and European levels, there is currently exploration of the possibility that "networks of long-term monitoring sites" could be used for comparative studies of the ecosystemic services (see for example the use of the sites of the International Long Term Site Monitoring system in the United States). **It is necessary to reinforce the participation of French teams in networks such as LTER Europe, by using and supporting certain work carried out within the framework of the "workshop zones" and "biosphere reserves".**

As the Operational Research Committee reminds us, it is not just the ORE that make up the observation system, but all types of platforms, sites, long-term systems, and above all their coordination in networks. The observation of biodiversity mobilises very diverse scales (from satellites to field observation) and organisations that are just as diverse (from the finalised organisation of nature inventories to the accumulated contributions of volunteer citizens). The world programme GEO-BON (Global Earth Observation-Biodiversity Observation Network), recently launched by Diversitas, offers the opportunity to think about the structuring of the national biodiversity observatory, as selected by *Grenelle de l'environnement*, in an ambitious manner, providing for the organisation of a continuum between the nature inventories, generally done by volunteers (base of the pyramid) and, at the other extreme, very high-level research on the functioning of ecosystems/socio-ecosystems, such as that carried out within the framework of the workshop zones of the CNRS. **Such programmes must work on including the socio-economic observations within their scientific systems.**

We see that the observation of ecosystems and nature is in the process of generalization. On the political level, Grenelle acted on the need for a national biodiversity observatory. **It is important to be able to associate within these information systems indicators of state and indicators of pressure on biodiversity linked to the various human activities. This concern ties in with the issue of the definition of relevant spatial entities – qualified as "socio-ecosystems" – for the description, analysis and management of biodiversity.** The initiative of MEA France that is now in progress could offer a response in this regard, as long as it is careful to take into account the socio-economic data in the typology that will be established.

This component was mentioned again by the operational research committee of *Grenelle de l'environnement*: "the ORE must be maintained and developed. These are research systems allowing for the coordinated measurement of different variables (…) as a function of the overall changes, either climatic, or those linked to human activity."

Lastly, the observation data are only of value to the extent that they are used. Support for public policies and the development of economic reference tools means also designing these observation systems as real tools for on-going development, making accessible to the public information organised for its understanding and its mobilisation in the public decision-making processes. **For the production of reference values, the purpose of these observations is the improvement of knowledge on the evolution of the value to be given to ecosystems as a function of their intrinsic evolution.**

## 2.4. Supplement the types of situations evaluated

### *a. Supplement the functions and services inventoried, particularly the contributions to health*

**Several initiatives around the world aim to deepen our knowledge of the services provided by ecosystems**: there should be active participation in projects such as RUBICODE (see chapter IV.3.5), which aims to get a scientific fix on the issue of ecosystemic services.

Certain areas of everyday life seem to have never managed to demonstrate certain forms of interdependence with biodiversity, while others demonstrate insufficient consideration.

**We can mention here the glaring lack of data connecting public health economics and the functioning of ecosystems**. While we see the negative influence of certain wetlands in the propagation of illnesses (malaria), we also note that for various reasons, sometimes to do with hygiene, the positive links of biodiversity with health, which range from the capacity of society to find active molecules to regulators of organisms that carry toxins, have received very little attention. Health and biodiversity is a very undeveloped subject. Furthermore, older services, which are assumed to be documented, are not substantially "monetarised."

Moreover, we can cite the **lack of documentation (in the research done by the group) on the economic translation of certain services provided by the forests**, for example on the regulation of runoff, whether from the standpoint of the resource or more significantly the risks (flooding, soil erosion, landslides). Historically however, very substantial work has been approved (RTM work in mountains for example, that covers vast areas of mountains that used to be eroded and that generated cataclysmic torrents).

This work thus noted the **weakness of the economic analyses relating hydrology and economics** (for example, the economic consequences of modifications of ecosystems leading to hydrograph modifications, sometimes with a direct impact on the production of energy).

In another field, the collective scientific expertise of INRA "Agriculture and biodiversity: harnessing synergies" (see Le Roux *et al.*, 2008), observes that **the services are only known "in experimental contexts that are often far removed for real agricultural conditions."**

Agriculture is not the only area in which the potential services are not identified. The economies of poor countries mobilise more intensely (and in relative shares of GDP) the services provided by their ecosystems than do rich countries, which have artificialised the means of production. In particular, **there must be development of research and studies on the direct and indirect monetary economic effects of the existence of protected and/or officially-recognised spaces.**

Moreover, while the landscape has been the subject of various studies mobilising innovative systems for the revelation of preferences, it has not been identified as an element that contributes to the monetary value of the associated ecosystems. This is part of the observed lack of methodologies to take into account the spatial organisation of the elements of ecosystems, while the works show more and more the impact of this organisation on the quality of the natural functionalities at work. We can cite as an emblematic example the fact that entomophagous insects, which depend on hedges, do not go beyond certain distances in fields. The "proper" functioning of this service linked to biodiversity (they are natural auxiliaries against crop pests) thus depends on a simple morphological property of the landscape: the maximum distance between the hedges.

**Research must further investigate the monetary translation of the functionalities of biodiversity associated with landscape structures.**

Lastly, the links between biodiversity and the health (in all of its components) of the populations must be studied, quantified, and translated into economic value, which could allow for the identification of non-negligible stakes with respect to the costs of the health policies.

### *b. Expanding it to "orphan" ecosystems*

For various reasons (cost effectiveness, private and public property, extension over the territory, etc.), there have been many studies of forests. Moreover, there are elaborate regulation and financing schemes for them. They were examined in the body of the text.

However, the data, systems and analyses available are less numerous for wetlands, riparian zones, grasslands, moors, complex milieus, etc. A list of milieus to be studied will be drawn up, with special attention paid to tropical ecosystems.

A priority would be to rapidly do work on one or several working groups (mobilising specialists) to propose reference values for:

- Wet lands (systems widely studied in the English-speaking countries but not common in the French context, and not often facing issues of valuation in situations of scarcity);
- All forms of grasslands and moors;
- The forest of Guyana.

This requires the analysis and transposition of research, but certainly also new field studies.

Moreover, soils, their dynamics and their biodiversity, have long remained the poor relation in research although they are necessarily at the crossroads of all the environmental flows, and supports of very important and diverse functions, services and productions. The changes of use of soils are very important factors of modification of natural functioning. Soils are involved in many decisions relating to economic activities and involving consequences in the long-term. Agricultural areas in France are losing 60,000 hectares per year. Research on the functioning of soils is thus determining for the use of the services linked to biodiversity.

### *c. Improving knowledge of the costs of engineering and reconstitution of milieus*

Without the group being able for the moment to make a statement regarding the relevance of these approaches for the establishment of generalisable economic values, it seems that approaches involving reconstruction, reparation and compensation costs are tempting for public decision-making when it wants to correct the lost services and let the developers find the least costly solution. They offer the advantage of corresponding to many of the immediate concerns of the actors, with levels that are more or less advanced according to the countries on the compensation markets and in the field of insurance. This would progressively establish references for the use of biodiversity through the law (as in the United States).

In France, economic actors, facing the outlook of the transposition of the directive on the environmental responsibility of companies, are awaiting general references with a natural orientation toward the choice of compensation, reconstruction and/or substitution methods. Along the lines of the scientific working groups currently working, **there should be an increase in the constitution of reference costs while maintaining, in their implementation, strong attention to the limitations of compensation or reconstitution approaches** when they are considered as first rank responses in the socio-economic calculation.

In this regard, it seems worthwhile to encourage the presence of French teams in scientific programmes such as REMEDE[1], a programme involving research for a methodological framework for the evaluation of equivalences under the two directives, “Environmental responsibility” and “Habitats”.

# 3. Developing the taking into account of factors of human pressure in models of biodiversity

## 3.1. Taking advantage of the development of quantitative models

Models are simplified constructions aiming to reproduce a process. The principle of modelling is to isolate the effects of certain factors and to select the processes to be taken into account, associating mathematical terms with them.

$CO_2$ discharge is “easy” to quantify: these flows are the causes of climate change. The scenarios use a translation of the variation of the impact factor ($CO_2$ emissions) with modelling of the impact itself: the evolution of the functioning of the atmosphere and the climate changes induced. It was possible to use these models to answer the economic questions.

With regard to biodiversity, many of the available indicators are state indicators. It is all the more complex to arrive at models when we realise that biodiversity is more the fruit of complex dynamic interactions than of states. Measurement of interaction is complex because it is rarely directly quantifiable. For researchers, it is a real challenge to understand (and above all to quantify) how human projects will make biodiversity evolve, and what are the evolutions and consequences of the pressure factors

---

[1] Programme of the 6th FPRD.

generated by human decisions and realisations or by other natural evolutions. After having approached some economic values by favouring, within the framework of this report, the visions of the associated services, we must be able to mobilise them in choice processes and alternatives.

Quantification is necessary for several reasons:

- When the same human pressure has antagonistic effects. This is the case of the intensification of agriculture, with greater pressure on cultivated land, and a potential slowing of deforestation. The qualitative result (the impact on biodiversity) depends on the relative importance of these two effects;
- When there is a choice between different ways of occupying land: urbanisation, agriculture, ecotourism, etc.;
- In order to facilitate the comparison of the outputs from these models with those from the climatic and/or socio-economic scenarios.

This quantification approach can be based on a certain number of initiatives in the field of scenarios:

- DIVERSITAS and other ESSP programmes (Earth System Science Partnership) develop networks and agendas for the construction of “regional and global scenarios of the impact of global changes on biodiversity and on ecosystemic services”, using new modelling methods: pairing of climate-vegetation models, response of biodiversity to overall changes, relationships between functioning of ecosystems and biodiversity, etc.;
- IPCC, GES, MA, ENSEMBLES consortium: socio-economic scenarios, in the field of available data on biodiversity;
- EDIT and GBIF: placing in networks of databases of the biological traits of species and their distribution.

Models play an important role in the following types of expertise:

- MEA (Millennium Ecosystem Assessment) and continuation of the MEA (Subglobal Assessments);
- TEEB (The Economics of Ecosystems and Biodiversity) – Use of biodiversity and ecosystemic service scenarios in order to estimate the value of biodiversity and the cost of inaction;
- Global Biodiversity Outlook 2 and 3 – Appraisal of the CBD of the current and future state of biodiversity (GLOBIO model (GBO2), summary GBO3).

**These models must be coupled with scenarios for pressures. It is then important to know how to transform public policies or infrastructures into pressures and thus inject their hypotheses into the models of evolution of biodiversity**, anticipate disturbances and associate them with the economic values (evolution of the services first of all).

## 3.2. Reinforcing the capacity of economic modelling to generate scenarios useful for public decision-making

To use reference values in a decision-making process, we must be able to elaborate and present differentiated scenarios. We must stress the fact that the models and scenarios must be accessible to citizens within the framework of the construction of public decisions.

**For the moment, the links between indicators of biodiversity and monetarisation are too tenuous or complex to be easily mobilised. This is an orientation to be developed.** This assumes mobilisable indicators and scenario contributions. One basic recommendation would thus be to resolutely develop the production of observation and modelling data, to include monitoring over time in order to develop possibilities for simulation, adjustment, prediction and measurement of uncertainties. This should open to all the possibility of adapting the objectives of biodiversity associated with a territory to all of the mobilisable public decision-making alternatives.

**There was even discussion of the need to develop an observation and modelling system representing for biodiversity the equivalent of observation and modelling systems for meteorology.** There could be the objection that the weather interacts with strategic and economic stakes that are immediately identifiable and profoundly linked to the geographic dimension: transports, agriculture, conflicts, etc., whence the immediate interest. In response, we must remember that the analysis of biodiversity and of the systems connected to it and also its value are already a determining criterion for the localisation of economic activities, not just for extraction (mines) or those that are potentially polluting, but also in conjunction with effects on the "living environment" of employees and the conditions of competitiveness and insurability.

We can also say that in the future, preservation of the environment or simply the needs of management (integrated pest management, dissemination of pollen, pests, collective strategy in agriculture) can also benefit from such monitoring and modelling systems on an operational level.

Lastly, among the clear necessities for the development of capacities for dynamic modelling of biodiversity, we should mention **the characterising of the relative price elements for biodiversity over time.** In issues relating to discounting, there was a reminder that, as for the ratio to a tonne of carbon, the taking into account of the future value is done with a homogeneous discount rate for all goods, but with relative price variations. These variations will not be the same depending on the elements of biodiversity under consideration and their evolution. Comparisons of alternative decisions on socio-economic appraisals require that we avoid excessive errors in terms of relative prices.

### *a. Developing indicators that materialise the expected effect of infrastructures and human activities on biodiversity indicators*

The report discusses the needs for the development of indicators identified in the new scientific strategy of the French Biodiversity Institute (IFB, July 2008). From a standpoint of public decision-making mechanisms, we should focus on characterising the impact of the decisions (infrastructures, developments, activities) on indicators of value, in order to move from a natural capital approach, or state of degradation, to a "marginalist" capacity for appreciation to evaluate the consequences of the various alternatives and to arrive at a real rational choice mechanism.

On this point, the developments on the indicators stressed the particular value of the so-called abundance indicators:

- There are certainly enormous possibilities for error when characterising the biodiversity of a milieu with these indicators (for example, population explosions can be a sign of serious biological malfunctions);
- Nevertheless, variations of abundance, integrated in a system of observation over time, act as a "sentinel" for pressures on ecosystems and biodiversity.

The sensitivity of the economic values obtained to the hypotheses made on the discounting must not obscure the importance of the uncertainties concerning the scenarios for the evolution of biodiversity and the ecosystemic services themselves.

With regard to scenario calculations, there should be research and re-examination for the use of discounting and the evolution of prices.

### *b. Exploring the extension of the field of the socio-economic calculation to the more ordinary decisions that affect biodiversity*

The methods of monetarisation are currently limited in the taking into account of biodiversity that we initially agree to refer to as "remarkable". The work of the group encountered this obstacle in the attempt to generate a comprehensive value. Moreover, many infrastructure projects for which the socio-economic calculation is mobilized are also remarkable (TGV, motorways) from many standpoints. However, **as the spatial planning decisions that have consequences on biodiversity are numerous and currently characterised by the under-estimation of the environmental services: infrastructures (roundabouts, exchangers, business zones, parking lots, loose urbanisation, etc.) are spreading at the expense of various more ordinary, non-developed biological spaces**. As for infrastructure, the integration of biodiversity with a socio-economic evaluation of day-to-day development policies should clarify the choices of alternatives.

An analysis of the efficacy of the reference values that can be arrived at would usefully clarify the systems and/or developments which the valuation of biodiversity would lead to renouncing or modifying or developing in the interest of the efficacy of public policies.

**There should probably also be experimental consideration of the conditions for application of the socio-economic calculation to new fields**:

- The evaluation of the green GDP, an action already in progress;
- The integration of the value of biodiversity in the first place in the criteria for the evaluation of acts confirmed by the State in relation to the preservation of biodiversity, for example in the exercising of its authority in the various territorial planning systems (SCOT, PLU).

## 3.3. Characterising insidious mechanisms to integrate them in the scenarios

The erosion of biodiversity begins well before its scarcity. In many ways, it is the small things that quickly lead to major difficulties, without each microdecision in itself leading to the anticipated catastrophe. The image of airplane rivets (Ehrlich and Ehrlich, 1981) is very illustrative in this regard. After how many lost rivets does the aircraft lose its wing and is it really that last one that protects it against accidents? Another analogy, in terms of natural phenomena, the insidious impermeabilisation of vast urban areas through successive building permits leads to flooding as in Nîmes: the answer might be an absolute obligation that each new construction be designed so that it does not modify the characteristics of the flow of runoff water. In the case of biodiversity, once the objective of stopping the erosion of biodiversity is announced, does this assume the intangibility of each ecosystem? Does the compensation, in whatever form, correct the "insidious" side of erosion by small steps?

As the debate on compensation grows, that of marginal pressures must not be forgotten. Thus it must be possible to provide the regulatory provisions requesting analysis of the impact of the various tax and regulatory systems on biodiversity with further knowledge.

## 3.4. Enhancing the conditions of transfers of value between sites

Transposing values obtained locally remains an extremely complex exercise, but this is the basis for establishing "reference" values. The manual of the OECD for use by decision-makers discusses the difficulties of this and specifies that the problem is currently far from being solved.

As specified in the appraisal section, Kirchhoff *et al.* (1996) empirically validated the fact that **the transfer of values in the form of function (of a set of independent variables) was more reliable than the average unit transfer of values for a certain type of site.** They also emphasised that the circumstances in which the transfers are valid and relevant for the decisions could be rather limited and that the errors resulting from the transfers could be considerable, even between sites presenting quite similar amenities.

The issue of the transfer of value is determining for the use of the values observed in the elaboration of reference values, particularly according to two points of view:

- **Spatially**: on the possibility of locally adopting a value from another situation, possibly corrected for certain functions of adaptation to the objective differences with respect to the references;
- **Temporally**: the socio-economic calculation used to enlighten public decisions. We must put values in the decision scenarios and know how they change.

This is only a part of the problems raised to arrive at a dynamic evaluation of the elements of biodiversity and to work towards models.

## 3.5. Determining the spatial limits of the transfer of average values and calculation mechanisms by establishing a socio-economic and biogeographic reference system in the territory

In compensation strategies in particular, the observance of pedoclimatic conditions is not the only factor: the sociological context can also be determining. The rate of visits (people walking, animals) of areas that are otherwise equivalent from a standpoint of biological potential can depend greatly on the proximity of cities. These considerations, currently determining in the debate on the validity of strategies of compensation, exchanges and "reconstruction" of ecosystems, are obviously reflected in **questions regarding the parameters to take into account in the monetarisation from values not calculated locally or simply on the method of calculation from reference values.**

Along the lines of the process implemented for the framework directive on Water for which the bodies of water were defined, it would be useful to mobilise the biogeographic reasoning or ecoregions already defined in the various public policy strategies.

## 3.6. Developing methodologies for the taking into account of the spatial organisation of pressures and elements of biodiversity

**For the designer of an infrastructure, it is obvious for example that the physical continuity of a road is an element that is determining for its value:** consequently, locally, the need for this continuity may lead to particularly high expenditures, which are very far from the average value of the infrastructure: a viaduct for example.

At this stage of the group's thinking, it was not possible to take into account the effects of the spatial organisation of the elements of the ecosystems in the values proposed. However, and especially for ecological "infrastructures", this thinking can sharply adjust the average values. It is obvious, for example, for the upstream travel of migrating fish and the effect of dams. It will become so for the new approaches of spatial policy for biodiversity applying green and blue frames. Some regional indicators (satellite images) can provide ideas of density and local dispersion, but they are very far from being able to characterise landscapes and their ecological functionality.

These topological approaches could provide elements useful for:

- Better taking into account of the extent of the impacts of infrastructures;
- Optimisation of the ecological interactions between elements of the landscape;
- Association of indicators of density, scarcity (developed in the text based on the inventories and the remote detection tool, particularly with the European Environment Agency).

The scope of a methodology of valuation of the spatial organisation of the elements of biodiversity is very important: beyond the establishment of results that are potentially valuable for the socio-economic calculation, this offers developments that are determining for the collective accounting of environmental assets and for their evolution: once again, insertion in works of international scope is indispensable, such as for example the methodologies explored in the parts of phase II of the TEEB.

### 3.7. Marking the limits of validity of the supposed effects

For both ecological and economic reasons, it is necessary to be prudent when we extrapolate and add up values estimated from marginal changes, of small size, to evaluate the effects of major changes. Ecosystems often respond to changes in a non-linear way (many programmes including for example EPBRS recommend attentively studying the "complex dynamics, non-linear responses and abrupt or irreversible changes"). Extrapolation of the benefits in the alternative scenarios must take this into account. Moreover, not all of the biodiversity functions of an ecosystem exposed to impacts are affected in the same way. For example, the biological production of a forest subjected to noise is not modified although its recreational value is: we could even consider that as the number of visits declines, the functioning of the ecosystems improves.

## 4. Economics and social sciences: deepening the relationships between economics and politics concerning biodiversity

### 4.1. Enhancing the analysis of the link between the political objectives on biodiversity, the economic analysis and the socio-economic calculation

#### *Translating the objectives of the ecological policies and making the methods reliable*

In the field of economics and social sciences, the working group encountered a double lack: of concepts on the one hand, and of proven methods applied to concrete situations.

Priority should be given to trying to better answer the following questions:

- What do the objectives linked to biodiversity mean for society?
- What are the objectives that society must pursue?
- How are they to be reached?

- *What is the economic meaning of the objectives related to biodiversity for society?*

From the standpoint of the economic and social sciences, setting objectives concerning biodiversity actually induces an additional scarcity in natural resources which we were just beginning to see as limited, but which are thus all the more so because of the objective of preservation. Thus, the strategies for ecosystems limit the available spaces and activities. With respect to pollution, it is the allowable substances and the services that can be expected from them that become limited.

- *What are the objectives that society should pursue?*

The question seems very general, but it concerns the bases of the social value of biodiversity and the means and methods that allow us to consider it and to clarify its determinants. This leads us to want to:

- Make explicit the links between measurements of biodiversity and the value of the services and, if possible, to move forward with the definition of metrics of biodiversity and landscapes;
- Develop adapted econometric methods: metrics accepting the heterogeneity of preferences;
- Promote work on an economy of biodiversity in a post-welfarist framework (it will probably be necessary to find a way to go beyond a utilitarian framework, even a sophisticated one, in order to match the evaluation and choice procedures with criteria such as freedom of choice);
- Incorporate the effects of the non-linearity of impacts, analyse the resiliency of the socio-ecological systems;
- Analyse the issues raised by invasive species: should this be done with a cost/benefit or a cost/efficiency outlook? If we consider that the impacts of invasive species are unpredictable, the preferable approach would be of the cost/efficiency type or more generally *As Low As Reasonably Achievable* (ALARA), i.e. without reference to the expected benefits;
- Study the level (geographic, actors) at which the value stakes are played out: biodiversity offers local services and a comprehensive asset value; the interconnection of these stakes (value for whom?) raises many questions (for relations between the agricultural sector and the rest of society, for international relations and, in particular, North-South relations), etc.;
- Correlatively, it is necessary to work on organising the information on the projects to avoid a cumulative degradation of biodiversity (keeping in mind that the marginal value of the conservation increases very sharply when the overall level of biodiversity decreases).

- *How to reach these objectives?*

For this category, many questions have been raised and are or will be the subject of priority work:

- Biodiversity appears as a set of mixed public goods, the original characteristics of which must be specified again, particularly with regard to preservation/production strategies. For invasive species for example, preservation is of the "*weakest link*" form, i.e. that the quality of the preservation is not as much influenced by the sum of the efforts as by their distribution. It is enough that one agent (or a small number of agents) does not make an effort for the efforts of the others to lose their efficacy. And we still know little about ways to manage efficient collective actions in these cases;
- To protect biodiversity, *in situ* protection actions appear to be essential, in both Metropolitan France and in the Overseas Departments and Territories. But there is much reticence and beyond the evolution of the legal framework (law on National Parks of April 2006), economic analysis can offer solid arguments to justify one project rather than another one. The association of national parks of France (PNF) and the MEEDDAT

recently had a study carried out by a consulting firm on the economic impacts of the Parks. Without going into detail on the results and limits of this work, it revealed the importance of research work in this field and, in particular, applied research work that would integrate the problems of limited information with an outlook of optimal selection of the sites to be protected. Such work was done recently, particularly in the United States (S. Polasky, references in chapter V);

- This search for efficiency in protection also involves the incentive dimension of public policies. In many countries (United States, Australia, etc.), the subsidies granted to farmers, foresters or other categories of holders of spaces and land, are allocated through competition procedures, particularly in the form of auctions where the public authority is in a monopsony situation (inverse situation from a monopoly; there is a lone applicant and many parties offering. The public authority is often in such a situation). Work in this area could be encouraged in France.

## 4.2. Analysing the economic values of biodiversity revealed in public policies

**The progress needed to generalise mobilisable shadow values in the socio-economic calculation currently depends on many unknowns in terms of monetarisation capacities: they require research and experimental studies with *ad hoc* accompaniment tools.**

In infrastructure calculations, the IRR (internal rate of return) contributes substantially to the debate on the relevance of the realisation (without however definitively constraining the political decision). This socio-economic calculation mobilises “shadow” values (value of time, etc.).

These shadow values provide values that are mobilisable for all of the public policies that impact the monetarised object. The modes of calculation and discounting have varied over time[1]. The values and the rates were finalised by the working groups at the cost of compromises and even conventions. For biodiversity preservation policies, the absence of comprehensive macro-economic monetarisation of all of the values of biodiversity does not currently allow for judging of the economic optimality of the overall objectives.

This difficulty has led firstly to the search for a conceptual framework for the evaluation of the net economic consequences of actions to address an objective of preservation of biodiversity and ecosystems. This framework must also be able to provide a link between:

- The more or less differentiable shadow values as a function of the spatial parameters;
- The estimation and localisation of the marginal changes of costs and advantages linked to these measures;

[1] Calculation conventions, particularly for the discounting rates of the various ratios of the General Plan Commissariat called “Boiteux ratios”, are very different.

- The possibility of integration of other "values" in the economic calculation beyond services alone.

The study *Review on the Economics of Biodiversity Loss: Scoping the Science*[1] was in particular analysed in the report *Economy of the ecosystems of biodiversity* (TEEB stage report): the appraisal shows that we are still far from being able to provide the bases of an explicit economic evaluation on the spatial level, based on quantification and localisation of the processes of generation of benefits for humanity from the functioning of biodiversity and ecosystems. Knowledge of ecosystems is very uneven.

For France, the spatial factor is particularly important:

- Metropolitan territory: crossroads of diverse biogeographic zones and highly anthropised ecosystems, in which the anthropic pressures also strongly affect the ordinary biodiversities;
- Overseas Territories: more specific ecosystems, insular, oceanic, "Amazonian", etc.

Considering that all of the research on these spatially characterised productions and benefits has still not defined them and that many key roles (thresholds, keystone functions) are not documented, and that various forms of value are not attainable by "econometric" means, but that we are not at liberty to wait to preserve biodiversity, **it seems necessary supplement the current knowledge on the values of services rendered by the study of the implicit values ("*shadow prices*"), revealed by past political decisions. The backwards calculation could apply to very many forms of political decisions involving biodiversity.**

In this way there can progressively be contributions to formulating for **each "socio-ecosystem", and for each service,** an established **historical reference value**.

The extent of the difference between the value of the services rendered and the value linked to the decision can help in analysing the scope and the limitations of the socio-economic calculations proposed in the report or to formulate indications for the economic taking into account of biodiversity in decision-making.

## 4.3. Developing analysis of procedures for the taking into account and sustainable management of biodiversity

Another important dimension in the mobilisation of social sciences involves the analysis of relevant procedures for the taking into account and sustainable management of biodiversity. We see these limitations in the case of the establishment of shadow values and the regulation of their use, but also in the issue of the expression of preferences in the case of contingent valuation. We can also consider this in the case of the reaction of a certain number of actors of society to various incentive systems that could be implemented, based on the evaluation of the ecosystemic services: it would be naïve to think that these actors will behave like "*homo œconomicus*"! This leads us to reconsider still further the question of the

[1] *Review On The Economics Of Biodiversity Loss: Scoping The Science*, Balmford A. and Rodrigues A. (University of Cambridge), Walpole M. (WCMC), ten Brink P., Kettunen M. (IEEP), Braat L. and de Groot R. (Alterra), May 2008 (for the DG Environment, European Commission).

procedures that would allow for the formalisation of value: can we combine the economic valuation and deliberative procedures such as juries of citizens or other systems? Various American projects in "real" conditions – because they were established on the occasion of actual situations (pollution, oil slicks, accidents) – turned out to be forms considered to be efficient at revealing the economic roles of ecosystems.

Does the conservation of biodiversity have a legal status that is relevant and that allows for effective protection? And how does it fit in with issues of economic value?

It turns out that only part of biodiversity (the genetic resources of domesticated species, protected species, remarkable spaces, etc.) has a real status that specifies the rights and obligations of public and private operators. Ordinary biodiversity on the other hand, whether for example herbaceous flora, the macrofauna of the soil and especially the micro-organisms of the soil and water, are considered the private property of the people who own or use the land. If, as we have amply underscored, this ordinary biodiversity is a major determinant of ecosystemic services, we can ask whether the legal sciences should modify its "status".

- At the crossroads of economy and law, many questions are now open in the field of property rights, and their possible dismantling or sharing, particularly with the outlook for the creation of "environmental easements".
- The services provided by ecosystems and the asset value of biodiversity depend on their modes of management and use. This is obvious for agro-ecosystems, and it is also true for forests and wetlands. This has important consequences for the implementation of obligation and compensation systems. The management choices are in fact choices on the composition of biodiversity.
- The interfacing of policies of conservation of biodiversity and protection of climates is not limited to ecology. Certain incentive policy mechanisms (States are sovereign on their own territories) are necessary and are indeed in place, for example through the REDD (United Nations Collaborative Programme on Reducing Emissions from Deforestation and Forest Degradation in Developing Countries (UN-REDD) which is the subject of controversies).
- Lastly, the production of information on biodiversity is an object of research: this concerns information on biodiversity as public goods, the risks of strategic bias, the contribution of citizens to the production of information (motivations, constraints, need for coordination). It is with the organisations closest to the field (especially when decentralisation "responsibilises" the regional levels in the pursuit of general political objectives) that there should be efforts to find the way in which the law and owners, social actors and networks, can come together to produce efficient information on biodiversity.

There remains lastly the issue of the taking into account of the heterogeneity of preferences with respect to biodiversity, whether the diversity of view points between the various actors present in the territory or the tensions that could exist between the local appreciations and those of the other stakeholders from outside the territory and who have other priorities. It would be superficial to say the least to translate this diversity by an "average preference" in the absence of consideration of law, and work at the interface between economy, sociology and political science could shed light on this problematic.

### 4.4. Studying the functioning of new forms of governance incorporating biodiversity

Objects linked to biodiversity are now being made concrete, their interactions are being determined, and there is experimentation. We are expecting a great increase in the possibilities of physical and digital modelling, evaluation methods, and knowledge of the pairings between biodiversity, services, risks, and health. The incorporation of these elements in current systems of governance, or the implementation of new systems, is a field of research and experimentation in and of itself.

Biodiversity and the way in which society takes it into account are already subjects for negotiation for the implantation of economic activities. The evolution and efficacy of the new systems of governance are in themselves unknowns that must be examined, analysed and anticipated. For example, research dedicated to the governance of new green and blue frames requires research that is finally as legitimate as that on the economic models of management and operation of natural resources such fish stocks for example.

## Bibliographic reference

Ehrlich P. R. et Ehrlich A. H. (1981), *Extinction: The Causes and Consequences of the Disappearance of Specie*s, New York, Random House

Kirchhoff S., Colby B. G. et LaFrance J. T. (1996), « Evaluating the performance of benefit transfer: An empirical inquiry », *Journal of Environmental Economics and Management 33*, 75-93.

# Chapter VII

# Towards reference values

Using the indicators, methods and hypotheses presented in chapters IV and V, which make it possible to associate a monetary value with some of the components of biodiversity, directly or in terms of ecosystem services, is it possible to define reference values for these entities or, more exactly, for the marginal variations, positive or negative, of these entities that are associated with human activities?

We will first rapidly review the meaning of the concept of reference value and the methods we use to define them, and then we will speak about the case of greenhouse gases and the reference value of carbon, in order to demonstrate the similarities and, above all, the differences with what we are interested in. We will then examine various methodological problems associated with the use of the monetary estimates that are available.

In a fourth section we shall, using some examples, look at the case of remarkable biodiversity in order to identify the limits of the economic approach in this area. The fifth section will examine more closely the concrete cases of the coral reefs, wet zones, forests and temperate zone grasslands to show up to what point it is possible to go in estimating reference values for these ecosystems. We will then look at the possible effectiveness of these values if, over and above the socio-economic calculations, they are used to influence a number of changes in land use.

The seventh section is devoted to procedural questions - the general problems of which have been presented in chapter III; that is, the approaches to be implemented for the setting and, above all, the use of reference values.

Finally, while, as we have indicated in chapter I, the setting of reference values in monetary terms is essentially intended to feed into and complete the *a priori* socio-economic analysis of public decision-making, the question of *a posteriori* treatment of the possible residual impacts of these choices also deserves to be examined. This will be the subject of the last section, on compensation.

## 1. Reference values and shadow values

### 1.1. Definition and terms of reference

As indicated in chapter V, **the concept of a reference value can be defined as a fixed value used by the public administration to take into account, and attain objectives relating to assets depending on public action and for which the value does not seem to be sufficiently appreciated by society or taken into account by private operators in the economic sphere. For such assets that are distinguished**

**as "shadow" (see V.1.2), we sometimes use the term shadow values to identify these reference values set by the public authorities.**

The necessity of such an exercise becomes apparent therefore when the functioning of the public contract, within the framework of existing institutions, does not correctly take account of a number of social and environmental costs and so these costs have to be internalised. **These values are intended particularly for inclusion in the economic assessment of the public investments that have to take into account, as far as possible, all the impacts, positive and negative, of these investments for the whole of society.** But they may also set up other regulatory instruments – taxes, quotas, standards, subsidies, etc. – the relative effectiveness of which must be estimated in each special case. **This important point, mentioned in chapter I, on the tools that can be used to include these reference values will not be discussed here** and we refer the reader to the recent interim report of the European Union (European Communities, 2008) in particular, devoting quite a long discussion to this question, particularly from the angle of an equitable sharing of the costs and benefits.

**This setting of reference values, even if it may be legitimate not to stick too closely to a strict utilitarianism, cannot however be arbitrary.** As soon as global public investment is limited, the reference values are there to facilitate the "best choices" between different possible spending options to be decided, aiming at the improvement of the "well-being"[1] of society. It is moreover advisable to make sure that the gains (or avoided losses) associated with the setting up of this value will effectively be greater than or equal to the cost overrun it will impose on operators in the economic sphere.

These terms of reference give rise to three observations.

## 1.2. The necessity of a precise objective

Given this definition, the setting of reference values *a priori* can only be done if the objective and the deadline for the public action have previously been declared. Thus, in the case of $CO_2$, the reference is the European commitment of 2007 to reduce emissions between now and 2020 by 20% relative to the 1990 value, with a perspective of a reduction of 60 to 80% between now and 2050[2].

In the case of biodiversity, two distinct *a priori* objectives can be considered.

1. Practise the truth on costs; that is, internalise in the calculation of costs, public (socio-economic calculation) or private, the negative effects on biodiversity. Even if, as we see, the estimating of these different effects is not always easy, this objective is sufficiently precise as a framework for the calculation of reference values.

2. Consider the current reference commitment of the national strategy for biodiversity; that is, to **stop the erosion of French biodiversity, including that of the DOM-TOM, between now and 2010.** Besides the problems mentioned in

---

[1] In the economic sense of the term; that is, including the satisfaction of basic needs.
[2] Following the Grenelle Forum on the Environment, France has itself committed to applying the "factor 4", the undertaking to reduce its emissions by 75% by this date.

chapter IV, of the representative and reliable character of the biodiversity indicators that can be used to verify that it has been achieved, this objective raises four main questions:

- **it is almost immediate and does not allow, as in the case of $CO_2$, the definition of "adaptation trajectories" and the possibility of being able to compare them.** As a corollary, it posits the hypothesis of the absence of ecosystems inertia; that is, the instantaneous effectiveness of the measures taken, whereas we know that some biological cycles take place over several years: for example, the restoration of the great European rivers will only see the significant return of salmon and other migratory fish after at least four to five years. Moreover, we may wonder even about its relevance to the climate change objectives: why is it necessary to act more quickly on the evolution of biodiversity (to stop it) than on climate change, when the inertial factors of these two processes are possibly quite similar?
- **it has not been clearly established that this objective is relevant from the ecological point of view nor optimal from the economics point of view.** Do we think that the erosion that has been occurring until now has not had clear damaging effects? Is it not now necessary to set a more ambitious objective to restore higher levels of biodiversity, in order to increase the ecological capital of our country?
- **it considers the national territory as an entity and does not give details as to how the accounting of the losses and gains can be done.** At what spatial scale will the variations in biodiversity be measured? Do we accept that the losses observed at a given point are compensated by the gains obtained at another point, whatever the national territory? Do we consider, on the contrary, precise territorial sub-units that have to satisfy the objective of biodiversity stabilization? As an example of this second option we can cite the idea of "water bodies", defined in SDAGE (water development and management master plan). These elementary entities – there are approximately 5,000 for the whole of France – all[1] have to attain the objective of being in a "good ecological condition" by 2015;
- **it does not raise the question of possible "externalities" of the measures taken to stop the erosion of biodiversity on the national territory.** It is possible that some measures that are appropriate at the national level have negative effects on other regions of the world and that France externalises to these regions the factors that erode biodiversity. This may be the case, for example, if some polluting agricultural practices are eliminated by recourse to imports from other countries that still employ them.

**So it would seem to be desirable, particularly as part of regular revisions of the sectorial action plans, to detail those elements of the national strategy that may strongly condition the reference values to be used. In particular, it appears to be**

[1] Except for a justified exception: some water bodies may be declared to be "highly modified" and must in this case attain a "good ecological potential"; the deadline of 2015 may also be put back, particularly for reasons of economic cost.

**important to define as soon as possible the territorial entities for the monitoring and management of biodiversity** and we will return to this point later.

### 1.3. The limits of market prices and assessments based on the preferences

Second observation, **to the extent that the setting of reference values cannot be based simply on declared or revealed preferences, the methods aiming only to measure these preferences must be used with prudence**, in particular when the preferences are declared. This is particularly true for estimates obtained from contingent assessment which, aside from their high degree of imprecision, only give information on what citizens are prepared to pay and not on what they would effectively pay – or on what it effectively would need to be paid – to prevent or physically compensate for damage.

It may however prove to be useful to compare these two ideas (willingness to pay and real cost of necessary investments) to judge, in particular, "the social acceptability" of the reference values. Similarly, the estimate of the spending committed to conserve elements of biodiversity "reveal" the price that society puts *de facto* on these elements.

This approach may sometimes highlight the large distortions between speeches and reality (Pearce, 2007): the studies conclude quite generally on the fact that the spending committed, or likely to be, is lower than the estimates from willingness to pay (WTP) obtained from contingent assessment, even if Pearce (*ibid.*) points out pertinently that the studies showing the reverse are possibly published less frequently.

So, Loomis *et al.* (2000) assesses the cost of restoring 120,000 hectares along a river in Colorado at 13.2 million dollars[1] per year, whereas the WTP of the inhabitants in question is estimated at a minimum of 18 million dollars. Similarly, the Swiss government is paying farmers around 880 €/ha/year for agro-environmental measures, whereas the WTP of the Swiss population for the protection of the biodiversity of agricultural zones of mountain rises to 3,720 €/ha/year [2] (Günter *et al.*, 2002).

However, care should be taken when using these comparisons as, if the estimates for willingness to pay per person (or per household) are quite homogeneous (within a range of 10 to 100 euros per year and per household, most often, which often makes the person questioned quite indifferent to how to answer), two factors will heavily influence the global estimate:

- firstly, the suggestion of a single donation, for example a gift to a foundation, compared to an annual donation. Whereas, according to conventional economic logic, a single payment should be at least ten times greater than the amount of a repeated donation (e.g. 25 times for a

---

[1] Methodology: in all the following cases, we quote the values given in the publications, without update. We have only converted Canadian dollars to US dollars and the various European currencies to the euro. On the other hand, we have converted the units to express as often as possible a value / hectare / year. When this value was expressed per hectare, we have used a discount rate of 4%, equivalent to dividing by 25.

[2] The WTP is about 200 euros per person (from 156 to 282 euros according to the populations interviewed and the solutions offered) and was converted to euros / protected hectare.

discounting of 4%), the studies show a ratio that is more often of the order of 5, which would radically diminish the value of the WTP;

- secondly, and above all, the size of the population considered as being likely to contribute, allowing the conversion of the individual WTPs to a value per unit of surface area.

Table VII-1 illustrates the result of the combination of these two factors for a study conducted in Great Britain on the WTP for keeping 5% of the tropical rainforest, according to whether it is a one-off or annual payment that is suggested, and whether only British citizens would pay or that their WTP would be extended to the citizens of all the industrialised countries.

**Table VII-1: Consent to pay (in dollars per hectare and per year) to keep an extra 5% of tropical forest**

| Population in question | Single payment | Annual payment* |
|---|---|---|
| British citizens only | 4 | 26.8 |
| All industrialised countries | 25 | 1 400 |

* Accumulation over 25 years.

*Source: Pearce (2007)*

## 1.4. The two options for the setting of reference values

Taking into consideration the preceding remarks, two options are possible for the setting up of reference values. We refer the reader to the recent report on the shadow value of carbon (CASE, 2008) for a more detailed presentation and we will restrict ourselves to a quick summary.

**The first option, called the "cost/advantages approach" aims to estimate the real costs that society will bear if damage occurs.** It is advisable to internalise these potential losses of reference assets; that is, to have the economic players take responsibility for their negative externalities. This is the option chosen by the Stern report on climate change that puts a value on all the damage resulting from global warming at between 5% to 20% (according to the scenario) of world total GDP between now and 2050. It deduces from this the price that a tonne of emitted $CO_2$ should be given to internalise this damage; 85 dollars for the case of a scenario of "laissez faire" where the damage would be maximised. **This approach therefore makes it possible to estimate the maximum spending that it would be legitimate to commit, from the point of view of economic efficiency, to prevent or compensate this damage.**

The report of the Centre for strategic analysis on the shadow value of $CO_2$ has criticised this approach, on two points in particular:

- **the "damage function", the link between the increase in $CO_2$ and damage, is only partly understood** and the damage may therefore be undervalued[1], whereas, strictly speaking, this value of the damage has to represent an estimate of what it is legitimate to internalise; that is, to have

[1] They may also be overvalued and so lead to unnecessary economic penalties.

taken into account in the decisions of the actors. There is a risk of undervaluing the reference value;

- **there is nothing to say that this reference value will effectively lead to a mobilisation of the actors so that the target may be reached.** There is, for example, a paradox, that the more ambitious the target (for example, limiting the concentration of $CO_2$ to 450 instead of 550 ppm), the lower is the reference value since the damage is itself reduced [1].

It's this same approach that is proposed in the interim report of the European Union (European Communities, 2008), the coordinator of which, the Indian economist Pavan Sukhdev, in Bonn recently, gave a figure of 3,100 billion euros / year, 6% of global GDP, for the losses in biodiversity between now and 2050[2]. In our case, this approach would amount to the assessment of the economic loss associated with the reduction of biodiversity and ecosystem services.

**The second approach, called "cost/efficiency", is based on an estimate of the value that needs to be set to incite the actors in question to adapt and so reach the required target.** These actors are diverse and their cost of adaptation is variable (according to their activities, it is more or less easy for them to reduce their emissions). So we really need to have economic models that are quite precise, so that the fraction of these actors that may adapt for a given reference value can be estimated and so set a threshold leading to a mobilisation that would be sufficient to reach the target.

The Centre for strategic analysis has chosen this second approach for $CO_2$. It has resulted in a proposal of 100 euros for a tonne of $CO_2$ by 2030, to be approached gradually from the current value of 32 euros.

In our case, the core of the data available can be used in the cost/advantages approach and so it is this that we will develop in particular. We shall nevertheless then look at the possibility of a cost/efficiency approach.

But first it is important to continue the comparison between the case of climate change and what we have to deal with, in order to highlight several important differences that mean that a specific approach has to be worked out.

## 2. Greenhouse gases and biodiversity: a comparison

**The first difference that we would like to draw attention to has to do with the complexity of our subject**. In the case of climate change, the central role of the greenhouse gas effect and how this effect is driving the increase in atmospheric $CO_2$ and other greenhouse gases is widely recognised. There are still some debates to be had to establish the equivalence relations between these other gases and $CO_2$, and to

[1] The paradox, however, is only apparent and results from the confusion of two levels of logic: the political objective of limiting concentrations to 450 or 550 ppm translates, implicitly, the greater or lesser degree of optimism of the decision-makers regarding the rigidity of behaviour and the level of the incentives needed to prevent serious damage. So it is logical that the shadow value should express this anticipation: if it is felt that a stabilisation at 450 ppm is desirable, it is because it has been estimated that it can be reached for a reasonable social opportunity cost, translated by a limited shadow value; if it is felt that behaviour inertia makes a target lower than 550 ppm illusory, the shadow value will be high, as it expresses the obligation of sustaining high costs.

[2] This figure is not to be found in the interim report but was declared at the report's public presentation.

be able to describe the form of the relation between the increase of $CO_2$ and damage, but the object, a "tonne of $CO_2$" is considered as a metric that is both simple and satisfactorily expresses the targets to be reached. Moreover, the atmosphere is essentially made up of only twenty or so gases and their contributions to the greenhouse effect are, as a first approximation, additive.

In our case, we have had to devote a long preparation to the presentation of the multiple facets of the notion of biodiversity, the importance of the interactions between its innumerable components and then envisage radical, and rather debatable, simplifications to have a metric available that is appropriate for economic analysis; that is, to be able to distinguish remarkable and "ordinary" biodiversity and tackle these two dimensions with different economic approaches. Even with these simplifications the subject is still very multidimensional and we will have to come back to this point later. Finally, if the direct link between greenhouse gases and the temperature of the globe is clearly demonstrated, we have seen that the link between the disappearance of certain components (populations, species) of biodiversity and the functioning of the ecosystems is still a subject of numerous debates.

**Beyond this complexity, a difference in nature also needs to be highlighted, having to do with the conventional distinction between variables of pressure and variables of state** (or response). Emissions of $CO_2$ are clearly of the first type: the volume emitted constitutes a single indicator (using the equivalences with the other greenhouse gases) of the pressures of an anthropic origin and the climate, quite well described by a small number of parameters (temperature and precipitations), constitutes a response variable. This situation therefore lends itself well to a cost/efficiency approach with regards to the determining variable, $CO_2$. Biodiversity represents, on the contrary, a response variable, the description of which may not, as we have just said, be limited to a small number of universal parameters. As for the pressure variables, there are a number (destruction of habitats, pollution, introduction of species, over-exploitation, etc.), not always easily quantifiable and the functions linking these variables to changes in biodiversity are much more complex than those linking the atmospheric content of $CO_2$ with the greenhouse effect. It is, moreover, for this reason that we have considered (see chapter V) that it was not possible to develop a cost/efficiency approach.

**The second difference is the "globalised" character of the atmospheric gases**, that makes the place where damage occurs totally independent of where the emissions take place and, inversely, means that everyone would benefit from progress made at any point on the globe. This implies that it is legitimate to prioritise aid to those economic actors who are best positioned to contribute to a reduction in emissions, wherever they are and whatever the nature of their activity. In the case of biodiversity, there is also a global dimension (particularly the possibility of modulating the effect of climate changes by the sequestration of $CO_2$) but numerous services have a local or regional use and the effect of their degradation would remain relatively localised. Similarly, the organisation of biodiversity at these different levels and the nature of the ecosystem services are also very variable from one place to another.

These first two differences have several important consequences:

- **the definition of a single reference value, valid for the whole of the national, indeed European, territory, is clearly a problem**;
- the question of where elements of biodiversity will be lost or gained cannot be avoided. **This question cannot be limited to its technical and economic aspects and also has a major ethical dimension, in particular when the areas of biodiversity erosion are already economically and socially poor**;
- the question of the transfer of value, that is, the use at a point of estimates obtained elsewhere, becomes central whereas it is, quite rightly, absent from considerations about $CO_2$. To take a crude example, we can cite the case of species that have been introduced that, while they contribute positively to ecosystem services in their natural environment, and they may even have a value for the natural heritage there, they may nevertheless prove to be harmful in another environment. So a reduction of the numbers of a species in its area of origin cannot be assumed nor compensated by the proliferation of this species in another area.

**A third difference, the fact that a tonne of $CO_2$ has to be evaluated in accordance with economic criteria, and so within a logic that is strictly utilitarian, and is not something having an intrinsic moral value, does not provoke any controversy** [1]. In our case, the fact that not only are there non-use values but that biodiversity has philosophical and cultural dimensions that would make an economic approach irrelevant, even inappropriate, is a point of view that is widely held. This is why we have fully discussed the pertinence of the economic approach in the third part and why the “manual of biodiversity assessment” of the OECD (2001) and the recent report of the CBD (2007) devote quite a few pages to this question.

Associated with this utilitarian approach, **it is entirely accepted that establishing a monetary equivalent for a tonne of $CO_2$, as with any monetarisation of the rest, is part of a logic of substitutability**, allowing for the possibility of exchanging a potential damage e.g. an increase in the concentration of $CO_2$, with any another asset, in a tacit logic of conservation of the “global well-being”. Even if it is recognised that the potential damage is serious and that there is a threshold for the “vital prognostic” of humanity, it is tacitly understood that we are still working within a perimeter where marginal changes are acceptable and negotiable. It is moreover for this reason that the cost/advantages approach can be defended, since the sums set aside as potential damage may be used by the public authorities to establish elements of well-being that are judged to be equivalent.

In our case, the question of the critical threshold, the level at which variations, even marginal, can no longer be accepted, is clearly being asked and even made official by the policy objective of stopping biodiversity erosion by 2010. This is especially the case for elements of remarkable biodiversity, and it’s particularly for this reason that we have considered it to be inappropriate to base ourselves on monetarisation for this case (see chapter IV). We refer the reader  to Turner *et al.* (2003) in particular for a discussion on this point.

[1] On the other hand, there is an unarguable ethical dimension in the debate on the degree of “acceptable” climate change, the distribution of the impacts, long term integration, and the setting of values in terms of the $CO_2$ content of the atmosphere cannot be limited to an economic optimum.

**The last difference that we would like to emphasise is related to the status of the asset that biodiversity represents**, something we have already mentioned in chapter V. In the case of the climate, the fact that it is a "perfect public asset", satisfying the criteria of non-exclusion and non-rivalry[1], is widely accepted; which, in all the countries of the world, justifies public, state intervention to try to control the change in climate, in the name of the "failure of the market" for such wealth. On the other hand, the fact that biodiversity is a public asset – according to the preceding criteria – cannot be defended in all cases[2], even if this correspondence is frequently made: it is in effect possible to appropriate elements of biodiversity (see the debate on biological resources and human life patentability) and the rivalry for the use of certain ecosystem services, whether this is the use of water or living resources, shows this (see chapter V). **It is advisable therefore to base requests for public intervention and the recourse to reference values on other grounds,** and in particular much more on the integration of negative externalities than on the theory of public assets.

**In conclusion, while recognising the importance and the quality of the work conducted on climate change, it would seem to be unavoidable that we distance ourselves from approaches that aim to measure and manage biodiversity with a tool similar to the tonne of carbon. A specific approach, that we are now going to describe, has to be developed.**

## 3. Methodological questions

Before looking at some concrete cases, we shall mention here four main points concerning difficulties encountered in the use of the various estimated values for ecosystem services. We shall also suggest some techniques for dealing with these problems.

### 3.1. The dispersion of values

This first point relates to the lack of precision of the estimates and the very great dispersion of the monetary values proposed, ranging from a few tens to several thousand of dollars per hectare per year. Even for a given ecosystem service, whether it has recreational value, a protective function or direct use value, the estimated values fluctuate considerably. For example, Bann and Clemens (*in* OCDE, 2001, p. 25) assess the recreational value of the Turkish forests at 0.1 €/ha/year while Turner *et al.* (2003) suggests a figure of 2,290 $/ha/year for English forests.

**It is however possible to identify the source of this great dispersion**. Effectively, besides the variations associated with the diversity of the methods used, the relative age of the studies and the uncertainties associated with each method, an important part of this variation comes from the diversity of the environment studied, in both the ecological and social sense of the term. **This is why, aside from the fact that each**

[1] Non-exclusion = it is not possible to exclude someone from the benefits arising from controlling global warming; non-rivalry = the fact that some benefit is not to reduce the possibility of benefit for others.
[2] In some cases, biodiversity is *res propria*, in others, a national and non-global public asset, in others a common asset (common pasture) sometimes, finally, a global public asset (open sea beyond 200 miles, the sea bed, the Antarctic).

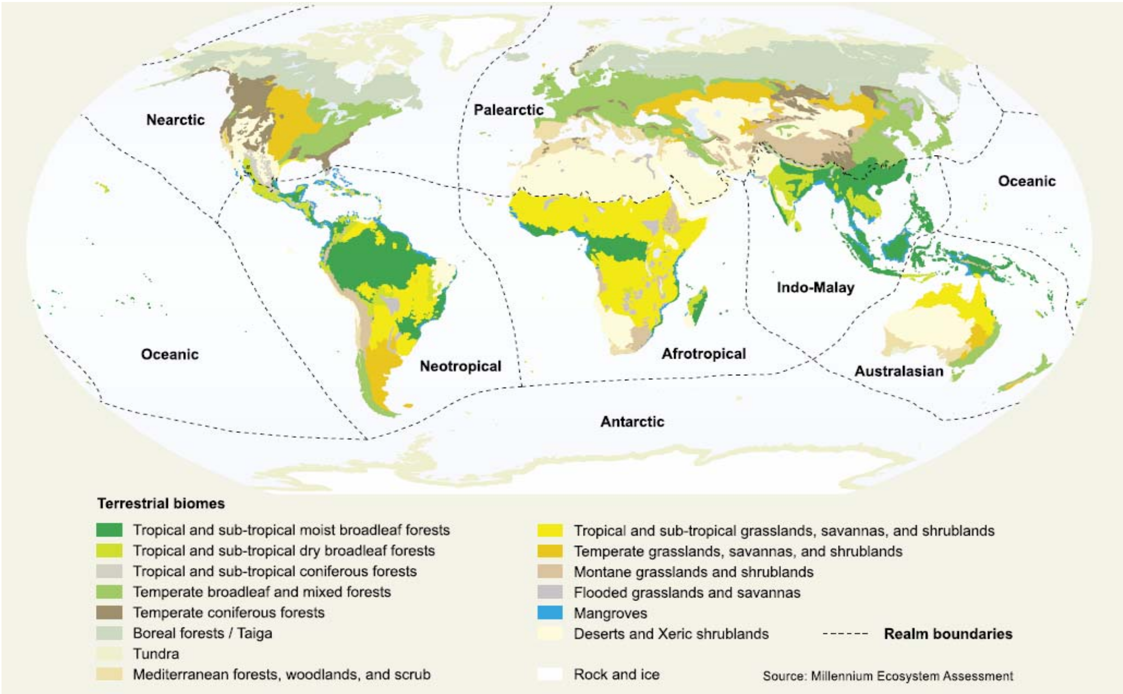
**service is examined specifically, it would seem to be indispensable to structure the analysis around "socio-ecosystems"**, that is, assemblies characterised by both their ecological and socio-economic specificities. These socio-ecosystems would therefore be defined *a minima* by four parameters:

- **their geographical zone.** There are at least, for the metropolitan region, four bio-geographical regions as defined in the Habitats directive, that is: Atlantic, continental, alpine and Mediterranean, and, for the overseas territories, the equatorial and wet and dry tropical regions. We see today in the studies a typology that is much more summary: for example, the study by Pearce (2001) on forests only distinguishes two types, "tropical" and "temperate", and the famous study by Costanza *et al.* (1997) uses the same distinction for forests but not for the continental aquatic environments over the whole of the planet. Similarly, altitude is generally not taken into account although its major ecological role is well-known;
- **the type of ecosystem.** There seems to be quite obviously a first net value gradation, from the cultivated zones with temporary vegetation cover to aquatic environments (coastal or continental) and wet zones, passing through permanent grasslands and forests. The Millennium Ecosystem Assessment distinguishes for the whole globe 14 great terrestrial biomes (figure VII-1) with a specific richness gradient (estimated in terms of the species of terrestrial vertebrates), ranging from the tundra to tropical forest and wet equatorial zones.

**However, within these great assemblies, a finer classification is to be developed, by limiting the reasonable number of types.** As an example, table VII-2 gives the European CORINE Biotope classification created in 1997, distinguishing 45 first level biotopes for the whole of Europe. The codification of the habitats used by the STOC programme (chronological monitoring of common birds), mentioned in chapter IV[1] can also be cited, identifying 7 main types of habitat, that may be broken down into 50 sub-types. Mention may also be made of two other European classifications: EUNIS, developed by the European environment agency, distinguishes 10 types of level 1 habitats (and a "complex of habitats" category) broken down into 57 level 2 types; EUR27, developed as part of the Habitats directive, distinguishes 9 habitats at level 1 and 35 at level 2.

[1] See http://www2.mnhn.fr/vigie-nature/.

Figure VII-1: Cartography of the 14 biomes selected by the MEA




*Source: MEA*

In accordance with the classifications, some habitats (for example forests or wet environments) are therefore described in a more or less detailed manner and the MEA France study in progress has in fact as an objective to propose a typology appropriate to our country.

- **the degree of anthropisation**, which may be approached by the population density in the region in question and that in part conditions the value of numerous services like recreational functions or the provision of high quality water. For this parameter, a classification at three or four levels (e.g. low = less than 10 habitants / km$^2$, average = 10 to 100, high = more than 100) would clearly be useful in the first instance;
- **the economic wealth of the country.** This economic parameter is not in the same spirit as the three other remarks made here, and the point has already been raised in chapter V but it may be useful to recall it here: the disparities of GDP between the countries investigated may induce strong disparities in the value attributed to an ecosystem. This factor provides, moreover, an explanation of the disparities observed.

**For a given large geographical zone, the idea would therefore be to have at most about one hundred reference "socio-ecosystems".**

Table VII-2: The 45 first level biotopes proposed by CORINE Biotope
(see http://in2000.kaliop.net/biotope/ibase.asp)

| Principle types | Biotopes |
|---|---|
| Coastal and halophilic habitats | 11. Seas and oceans<br>12. Arms of the sea<br>13. Estuaries and tidal rivers (subject to tides)<br>14. Salt licks and sand banks without vegetation<br>15.Salt marshes, salt meadows (salt marshes), salt steppes and thicket on gypsum<br>16. Coastal dunes and sandy beaches<br>17. Pebble beaches<br>18. Rocky coasts and sea cliffs<br>19. Islets, rocky banks and reefs |
| Freshwater aquatic environments | 21. Lagoons<br>22. stagnant fresh water<br>23. stagnant, brackish and salt waters<br>24. Running water |
| Heaths, scrubland and grasslands | 31. Heaths and scrubland<br>32. Sclerophyllous scrubland<br>33. Phrygana<br>34. Dry chalk steppes and grasslands<br>35. Dry silicate grasslands<br>36. Alpine and sub-alpine grassland<br>37. Wet grasslands and tall grasses<br>38. Mesophilic grasslands |
| Forests | 41. Caducifoliated forests<br>42. Forests of conifers<br>43. Mixed forests<br>44. Riverside forests, forests and very wet thicket<br>45. Non resinous evergreen forests |
| Peatlands and marshes | 51. High peatlands<br>52. Peatland cover<br>53. Waterside vegetation<br>54. Low-lying marshland, transitional moor and springs |
| Continental crags, screes and sands | 61. Screes<br>62. Continental cliffs and exposed rocks<br>63. Permanent snow and ice<br>64. Continental sandy dunes<br>65. Caves<br>66. Groups of volcanic sites |
| Agricultural land and artificial landscape | 81. Improved grasslands<br>82. Cultures<br>83. Orchards, coppices and plantations of trees<br>84. Rows of trees, hedges, spinnies, hedgerow, parks<br>85. Urban parks and large gardens<br>86. Towns, villages and industrial sites<br>87. Wild land and wasteland<br>88. Mines and underground passages<br>89. Lagoons and industrial reservoirs, canals |

*Source: http://in2000.kaliop.net/biotope/ibase.asp*

## 3.2. The integration of ecosystems dynamics

The second methodological point is associated with the question of “deferred” use values; that is, the potential services that an ecosystem may provide but which are not used at a given moment in time. This may be associated with the dynamic of the ecosystem: So, a young forest has weak carbon sequestration activity, even negative if it is planted following the exploitation of a mature forest (Magnani *et al.*, 2007). But this forest represents an important long term potential for this service. It is therefore as difficult to justify applying the maximum value for this service as to be limited to its current low value, particularly as the time necessary for the deployment of this service may be long.

These fluctuations may also be due to the dynamics of human activities. So the recreational value of an ecosystem may change according to different parameters (cost of transport, urbanisation, presence of substitutes) and it is legitimate, bearing in mind the time to set up this ecosystem, to take into account this value of potential use. The same reasoning may be applied to an environment degraded by various types of pollution but for which restoration is feasible. Taking the example of the great French rivers, it can easily be seen that the improvement of the quality of their water in the last thirty years has considerably increased services, whether in terms of biodiversity or recreational value: could Paris-Plage have been envisaged when the Seine was covered with a yellow foam due to surfactants?

To take account of this difficulty, we suggest establishing and combining two estimates:

1. **for each “socio-ecosystem”, and for each service, a reference value corresponding to the instantaneous value of this service.** The instantaneous value may be estimated, as we will see, by different means depending on the service. It may be adapted to a special ecosystem by using, for example, indicators of the “health” of the ecosystem, such as those used to assess the ecological state of bodies of water or the “leaf deficit” for forested lands. We can also work from a number of spatial-temporal parameters (maturity, heterogeneity) for the ecosystem to establish this value;
2. **similarly, a “maximum plausible value” that such a system may have in the medium term (30 to 50 years) in the geographical zone being considered.** This notion of maximum plausible value may be derived from a meta-analysis of the available bibliographical data or an expert view and possibly completed by prospective approaches. It may be revised regularly with the acquisition of new knowledge. The estimation of this value leads to including the inertia of the system; that is, its evolution that may be potential, spontaneous or associated with human activities. It is, for example, quite easy to envisage the ecological structure of a young forest in 30 to 40 years time.

**This notion of maximum plausible value is different from that of “maximum sustainable service”, which would be the highest value observed (or predictable) for a given ecosystem and compatible with its permanence.** As we shall see, these maximums (e.g. in terms of tourist numbers, carbon sequestration, purification of the water) may be much higher than the instantaneous values observed at a given place but there is no chance of their being attained at this place within 30 to 50 years (it is for example unrealistic to imagine that the Finnish forests may, on this timescale, be as visited as the forest of Rambouillet today). This is why the group suggests this approach to take the long term into account.

It may seem difficult to combine these two concepts of instantaneous value and maximum plausible value, but this really amounts to estimating the "average arithmetical value" [1] of this service over a period of about thirty years, integrating at the same time, as we have mentioned, the "natural" dynamic of the ecosystem and the possible changes in human activities[2]. So, the study of Magnani *et al.* (2007) on the carbon sequestration according to forest age leads us to suggest, for the European boreal and temperate forests[3], a value of 0.56 as coefficient to apply to the highest observed values to obtain this average value. The question is more tricky for other services, e.g. the recreational value or the provision of drinking water: a forest today that is seldom visited may, if urbanisation increases or lifestyles change, be in greater demand and it is not always easy to anticipate these sorts of developments.

## 3.3. The problem of summing values

Having obtained reference values for each socio-ecosystem and each service, we may legitimately wish to accumulate them to obtain more global estimates. This question has several levels:

- we may wish, for the same service, to accumulate isolated estimates. For example, having obtained values for the protection of various emblematic species, we may wish to obtain a global value for all remarkable biodiversity of a region;
- we can also, for a given ecosystem, accumulate the different services to obtain a global estimate for these services;
- we may, finally, wish to accumulate for a given territory the values of the services rendered by the various ecosystems of this territory.

**It seems quite natural to simply add the values together to obtain this global estimation, money being an additive metric. This is what we have done in the preceding chapter and it is also what most studies do, generally without discussing to what extent this approach is appropriate.** We can cite the study of Pearce and Pearce (2001) for temperate and tropical forests, proposed by the Czech Republic to the OECD to assess the whole of its territory (Seják *et al.*, 2002) or that produced by Costanza *et al.* (1997) for the whole planet. This option is also to be found in the concept of "total economic value", proposed initially by Pearce and Warford (1993, *in* OECD, 2001) and widely taken up, suggesting the simple addition of use and non-use values, whether current or future. But this really should be examined more carefully, as it does raise a number of problems.

**Firstly, it means adding the results obtained by very different methods** and placing at the same level observed prices (on a real market), revealed prices (we speak

[1] Possibly by weighting the values recorded at different periods by the discount rates.
[2] These two aspects, ecological and social dynamics, call on different expertises but it is interesting that a debate take place to develop a common vision rather than proposing a pre-established formula for combining them. We will return to this point when we discuss the procedures for deriving reference values.
[3] Twenty forests between 42° et 62° latitude North.

generally of "revealed preferences", via indirect methods such as hedonic prices or displacement cost) and prices (preferences) that are declared (when the contingent or "c*hoice experiment*" assessment is used). Now, if the observed prices do obey an additive logic, a number of studies have shown that this is not necessarily the case for fictitious prices, in particular those derived from contingent assessment. Several factors contribute to this result and, in particular, the phenomenon of "inclusion bias", described in chapter V. Thus, Boman and Bosdedt (1995 *in* OCDE, 2001) obtain an estimate of 126 dollars per year, per household, for just the protection of the wolf in Sweden, while Johannson (1989, *in* OCDE, 2001) arrives at only 194 dollars for the protection of 300 endangered species in this same country. Similarly, Hageman (1985, *in* OCDE, 2001) has only 18 dollars per year per person for the preservation of a number of endangered species in the United States, while values of 20 to 30 dollars are often obtained for a single emblematic species (see V.4.1). These examples also illustrate the poor perception by citizens of "ordinary" biodiversity.

**Some uses may also be assessed several times in a breakdown of services,** if, for example, the typology proposed by the Millennium Ecosystem Assessment is followed. Thus, the idea of recreational value, composed of "cultural services", can cover the practice of hunting, fishing and gathering, that will also be evaluated as "extraction services". This is the problem of "double accounting" (see Turner *et al.*, 2003).

**Next, the various services are considered on very variable scales**: some would essentially be used locally (e.g. the biological quality of the soil and its role in productivity), others are of regional use (recreational values, water quality), others still are to be considered globally, like $CO_2$ sequestration. By remaining at the local level, some services may be of benefit to the whole population, e.g. drinking water quality, while others, like hunting, only interest a special group. So special importance can be given to services of the first type. Finally, in some cases, there are even use services that are essentially non-local, for example when a operation to protect biodiversity is considered negatively by the local population. We can also cite degradation of the quality of running and coastal waters: the beneficiaries of improvement measures are principally located downstream of the places where these measures are taken.

**The precise definition of the populations in question is not always very easy and may sometimes reveal paradoxes**: several studies show, for example, that willingness to pay on the part of persons residing a long time in a territory is lower than that of persons visiting occasionally, even those, much more numerous, who do not visit but are nevertheless prepared to attribute non-use value to it. These questions are particularly crucial in the countries of the South, where one may be faced with poor financial capacities of the local populations for the food-producing resources that are nevertheless essential, confronted by a considerable investment capacity of actors of the North for another use of the territory, that may sometimes be harmful to these local populations (there are also positive cases, such as the development of ecotourism and fair trade). This complex debate clearly has an ethical dimension and may not be judged according to monetary arithmetic.

**This absence of additivity may be found, but for other reasons, when an accumulation of the value of different ecosystems on a territory is sought.** In effect, the ecology of the countryside clearly shows that the spatial organisation of the various ecosystems on a territory play a major role in the functional properties of this territory: according to their localisation and their proximity to other ecosystems, a grassy belt, a hedge or a coppice will play very different roles, in terms of the combat against pollution, protection against erosion, conservation of biodiversity or even the cynegetic or aesthetic value of this countryside. The problem of double accounts arises here also: the quality of the water from a forest ecosystem may, for example, be considered as a regulatory service for this ecosystem (see VII.5.3) and also be attributed to the services of aquatic ecosystems, as recreational services.

**More globally, we discover here the limits of an approach by ecosystems in composite territories and strongly anthropised as are the majority of the European territories.** It would be advisable therefore, on the basis of concepts of countryside ecology, to develop more global approaches that take into account the diversity and complexity of these interactions – ecological and also socio-economic – within a territorial mosaic and their consequences on the ecosystem services of this territory. Very concretely, this is a short-term question if we wish to develop an economic approach for the green and blue frame; that is, to estimate the plus-value associated with the linking up of previously isolated ecosystems.

**Finally, some services may be judged to be more important than others in a given context** and the arbitration between possibly antagonistic functions cannot result in a simple summing of the parts. So, we know that a fast-growing young and species-homogeneous forest may be more efficient, at least in the short term, in terms of $CO_2$ sequestration than an old and heterogeneous forest, but the hierarchy is reversed for biodiversity. Must the choice be made simply on the basis of the sum of these two services? Similarly, at similar total economic values, some environments may possess a relatively balanced “portfolio” of the various services and others may be characterised by a high value of some special functions; there may be reasons for choosing one rather than the other.

**This is why it seems necessary to us to also develop multi-criteria assessment methods, making it possible in particular to reveal and debate the weightings that the players attribute to these various services.** This multi-criteria assessment also means that qualitative and semi-quantitative information can be taken into account, and also the lack of precision of the various estimates, the different scales used and possible duplication in counting. Moreover, if this assessment reveals identical weighting, it provides an *a posteriori* legitimising of the recourse to a simple sum. We refer the reader to the OECD manual (2002, p. 71-74) for more details on these methods.

## 3.4. The standardization of the values by unit of surface area

As we have seen previously, numerous studies on the evaluation of ecosystem services, after estimating the services provided by the entirety of the zone considered (a forest, a lake, a moor, etc.), then convert the values obtained on the surface of the zone to obtain an estimate of service per hectare of surface area.

This has the advantage of facilitating comparisons between the same ecosystems in different contexts (e.g. forests that are more or less distant from urban areas), and indeed between different ecosystems (for example forests and permanent grasslands) and this is why we have used it in the preceding chapter.

But this simple metric has to be used with three important corollaries taken into account:

- **the hypothesis of a strict proportionality between the size of an ecosystem and the total value of the service rendered is not always legitimate;**
- **it is advisable to distinguish the physical surface subject to modification and the effective zone of influence of this modification;**
- **the localisation of the area subject to modification within the ecosystem may alter, for equally modified surfaces, the degree of variation of the ecosystem services, and to a great extent.**

*a. Are the ecosystem services proportional to the surface area of the ecosystem?*

In a relatively homogeneous ecosystem, we may find we admit that some services vary in direct proportion to the surface area of this ecosystem: this is the case with production services – hence their usual assessment in terms of yield per hectare – or certain regulatory services, such as carbon sequestration or functions that purify effluents. But other services work under more complex laws that are more or less well known:

- the biodiversity conservation function can often be understood, particularly for the large animal species, in terms of **threshold effects**: for large surface areas, there is an appreciably linear relation between the size of the ecosystem and the population size of the species but, below a certain minimum surface area, the accommodation capacity of the environment is cancelled out. Moreover, the genetic variability of a population, governing its long term viability, decreases in accordance with a non-linear law of the type $V = Vmax\ (1 - 1/KN)$, a function of N, the size of the population, and so a function of the surface area of the ecosystem, **which means that the same reduction of surface (in absolute terms) will have a much greater effect on small ecosystems;**
- **other aspects of biodiversity conservation are linked to the existence of interfaces between two environments**, called ecotones (edge of a wood, edge of a lake), constituting the zones sought by certain species, sometimes for a particular period of their lifecycle. This accommodation function is proportional to the scope of the ecosystem and so obeys, for an ecosystem with a form that is not modified but the surface is reduced, a law of the type $F = KS^{1/2}$, where $F$ is the ecological function and $S$ the surface area of the ecosystem, and so the value of the function will decrease less quickly than in a proportional function. Similarly, linear infrastructures (hedges, bands of grass running along water courses) owe their efficiency, beyond a minimum width, much more to their length and their continuity than to their surface area. We have such an indicator for French forests with the measurement of the length of edge per hectare of forest: this was an average of 50 m/ha in 2004, if we limit ourselves to the interfaces with other non-forest ecosystems but can reach 70 to 90 m/ha if the interfaces between different types of forest stands (populations) are included (MAP, 2006). This global indicator increases from 0.4% to 2% per year, but its interpretation is complex as it includes both a positive aspect – the increase of the ecotones – and a possibly negative aspect – the

increasing fragmentation of the ecosystems. It is therefore advisable to apply finer analyses to define its optimal value in different contexts;

- finally, for cultural services (tourism, aesthetic value), we can suppose both the existence of a threshold effect (the environment is no longer appreciated below a certain minimal surface area, with exception made for remarkable sites of course) and saturation effects (visiting by tourists does not increase once the surface area exceeds a certain size).

### *b. Physical surface area and the zone of influence of a perturbation*

For some services it is acceptable to consider that the surface area affected by a modification of the ecosystem is to all intents and purposes the same as the physical surface area effectively modified. For example, this is the case, as discussed previously, for production services or for carbon sequestration. Some aspects of biodiversity, for example that of soil micro-organisms, are also examples of this case. However, sometimes the impact may affect much bigger surface areas. An example of this is the case of transport infrastructures (roads, railways, airports), the "externalities" of which (air pollution, effects on the volume of tourism, reduction of the hedonic values of the habitations) should be taken into account on much bigger reference surfaces. To take an extreme example, degradation of the aesthetic value of the countryside by a high-rise construction would be considered for the whole of the area where this construction would be visible, a principle also applied to define the scope of protection of historical monuments and the coastline. Similarly, it is clear that nocturnal light pollution, and its disturbing effect for numerous species of bird, would not be reduced simply to the surface area on the ground around the streetlamps!

### *c. The importance of the localisation of the affected area*

**Even in apparently homogeneous ecosystems, the same modification, in terms of surface area, could have very different effects depending on its localisation.** So, linear infrastructures (roads, canals), fragmenting the ecosystem and reducing the possibilities for exchange between sub-groups, may have major impacts, for example on the populations of large mammals, while a sample of the same surface area but at the periphery will have much more limited consequences.

It is similar for all the processes governed by the "percolation laws", such as the propagation of forest fires. For the same de-wooded surface, this propagation will be more or less slowed down depending on the location of these clearings.

Inversely, the linking of ecosystems by linear structures having a small but judiciously positioned surface area may have a considerable positive effect on the whole complex of these ecosystems. This is the very question, already mentioned, which needs to be asked about the possible economic analysis of the green and blue frame.

So, if the analogy with transport infrastructures is used, the question can be asked not in terms of opportunity that is absolute but related to alternatives that are more or less costly and have a more or less satisfactory "performance". This concept of performance will of course include ecological considerations (capacity of the different species to travel within the frame) but, as soon as the costs are expressed explicitly, it seems that they should not be put on the scales with advantages, themselves evaluated economically. The workgroup has only touched on this question, one that deserves to be looked at more thoroughly.

**Moreover, this hypothesis of homogeneity of the ecosystem is not generally valid in reality**: there are always, if only for topographical reasons, “sensitive points” the modification of which may have major consequences. This is particularly the case for protection functions (work to combat erosion, protection against flooding or avalanches, preservation of springs), for which detailed mapping of the territories is needed to be able to judge the pertinence of average reference values, even those created for this type of ecosystem.

## 4. The case of remarkable biodiversity

We have suggested in chapter IV, in an ecosystem we should distinguish the elements of remarkable biodiversity and “ordinary” biodiversity.

At an ecological level, this proposal may be based on a number of existing devices: In France we have tools with which some of the remarkable elements of its natural wealth can be appreciated and mapped, in particular, the inventory of the Natural Zones of Ecological, Faunistic and Floral Interest (ZNIEFF). This is being updated for the whole of Metropolitan France and the Overseas Departments, and also the zones designated as part of the European network, Natura 2000. Also, the lists of protected species and habitats at the regional, national and/or European level, as well as those identified in the red lists of the UICN, give us the reference elements for the characterisation of some of the aspects of this remarkable biodiversity.

At the economic level we have questioned whether we should have automatic, and exclusive, recourse to economic analysis for this first case. Chapter V provides a series of arguments to justify this position. We are going to rapidly recall these elements, then back up this point of view making use of the lessons that may be learned from two cases of remarkable biodiversity, that of emblematic animal species and that of plants having pharmaceutical value.

### 4.1. Difficulties in the economic assessment of remarkable biodiversity

Several elements that constitute the conceptual framework for economic assessment combine to weaken the suitability of this approach for the entities that we have called remarkable.

### *a. The question of substitutability*

Of the three levels that we have identified in the economic concept of substitutability (see IV.1.2), only the third, the possibility of finding compensation at a psychological level (elements of well-being judged to be equivalent) applies *a priori* to the elements of remarkable biodiversity. It is a less robust basis for assessing technical substitutions, as the only mode for estimating these psychological compensations involves the measurement of declared preferences.

### *b. The measurement of existence values*

The elements of remarkable biodiversity may generally be characterised by the relative importance of non-use values in the foundations of their social value, compared to use values, mainly recreational or aesthetic. We have seen that the economic status of these non-use values was less assured than that of services that are more directly involved in the well-being functions of the agents. What is more, the very concept of remarkable raises the question of the population concerned that may potentially be very large.

### *c. Specific difficulties in discounting*

The idea of “remarkable” is equivalent, in a number of cases, to the idea of rarity, which increases the irreversible or irreplaceable character of the destruction. In the section relating to the specific features of discounting, we have emphasised that the irreversible effects of the destructions have to be translated into increasing relative prices (including the fact of option values and “extinction debts”) which are contrary to the effects of the discounting, and result in undefined and potentially very high values (if the effects of the losses are not temporary).

## 4.2. Some specific elements

### *a. The emblematic animal species*

Let us first remember that this idea of emblematic species is not only associated with ecological aspects. It also makes reference to the socio-cultural context and so may change quite rapidly from generation to generation. The change of "status” of the lynx, the bear and the wolf, hunted in former times, and today protected as part of the natural heritage, is without a doubt not entirely due to their rarity, already well-established, and illustrates this phenomenon very well. By limiting ourselves to the modern epoch and the western cultural context, we have presented in chapter V a number of studies of the populations of endangered vertebrates, for which we have estimated the willingness to pay, per household and per year[1] for different measures of protection for these species.

In the course of the application of the approach, these assessments have introduced several methodological refinements that do not however resolve all the problems in

[1] These are random samples of households, with no distinction between those that may use these resources (use values) and non-users, which would express the non-use value.

the use of these values as reference values. Besides the problems of additivity previously mentioned, the use of these values often raises the question of disproportion, and even disconnection, between localisation and the sometimes restricted character of the territory to be protected[1], the fact that these species are endangered, the importance and localisation of the population considered likely to pay, due to the emblematic character of these species, and the importance of their option or non-use value.

While for the elements of ordinary biodiversity we may assume that it is mainly those users living or visiting the site who are both those that benefit from these services, in terms of the use values, and those who are likely to finance the costs of management, these two populations are in fact virtually disjoint – and even sometimes antagonistic as we have already pointed out – when it comes to the question of remarkable biodiversity; which introduces into the debate undeniable political and ethical dimensions. It is therefore advisable, *a minima*, to develop approaches specifically identifying the use values e.g. in terms of local economic effects associated with the various labels awarded to elements of remarkable biodiversity (National parks, French "Grands sites", UNESCO World Heritage, etc.).

Also, for an order of magnitude, it would not be shocking to suggest that the number of people that would be willing to pay ten dollars per year for an endangered species of large mammal is of the order of a hundred million over the whole of the Planet. If the territory to be protected is of the order of several hundred $km^2$, we arrive at a WTP per hectare per year greater than 10,000 dollars, which, for a single species, would exceed all the estimates based on ecosystem services and would imply that all human activities that may affect this territory should be opposed.

This argument seems to be decisive and would necessarily lead to the "sanctuarisation" of the territories in question, but it has an Achilles heel: if the same citizens are asked if they believe it to be legitimate to spend a billion dollars per year of public money to protect the said species, the response is likely to be quite forthright! It would seem therefore to be desirable to replace these studies on a single emblematic species by more global approaches, presenting different programmes of conservation for a number of species to be protected in a given environment.

**For these different reasons, we feel it is preferable only to use economic analyses as supporting arguments in such situations and to confront instead the issues in the political debate in terms of all its dimensions.**

### *b. Plants with pharmaceutical value*

One of the oldest and most well-known examples of the exploitation of remarkable biodiversity was the agreement signed in 1991 between the pharmaceutical company Merck and the National Institute of Biodiversity of Costa-Rica: with a payment of 1.1 million dollars, Merck obtained for two years (the contract was then renewed in 1994 and 1996) the exclusive rights for the exploration and realisation (in the form of

[1] This is not the case for large animal species that, even at reduced numbers, still have significant requirements in terms of habitat. But the problem of "disconnection" is also raised.

rental charges) of the pharmacological properties of the plants and micro-organisms of the 105 National Parks of the country[1].

Around this question of "bio-prospecting" a number of studies have tried to estimate the medicinal value of a plant, not yet tested, that can be considered the potential value of an element of remarkable vegetal biodiversity. We shall summarise here the conclusions of reports of the OECD (2001) and of Pearce and Pearce (2001): they show that the value assigned to vegetal bio-prospecting varies from author to author, from 200 dollars to more than two million dollars per plant.

The reasons of this variation are to be found in two main factors:

- the probability of finding a valuable plant amongst all the plants tested varies from author to author, from more than one chance in 100 to one in 10,000;
- the value assigned to a plant ultimately used for a medicine; that is, the annual profit associated with this medicine, is estimated at between 250,000 and… 37.5 billion dollars. One of the main causes of this variation is linked to the fact that the medicine is considered either only in terms of private economics (the profits for the company) or has elements of public spending included (in particular the valuation of the number of lives saved).

In mathematical terms we are close to what is called an "indeterminate form"; that is, 0 x •. This means that the value of promising zones in terms of biodiversity will be estimated within a range between less than 1 dollar (which was the case with Merck) to almost 10,000 dollars per hectare.

**As we have already said, the use of these sorts of figures to defend the preservation of biodiversity would seem to be problematic. But the perfectly valid remark can be made that this uncertainty is characteristic of all pharmaceutical innovations, whether they are based on the exploration of biodiversity or other methods (combinatory chemistry, molecular modelling, etc.).**

## 5. Socio-economic analysis of some national ecosystems

We will here only look at some cases, to outline an approach that should be extended to all the national "socio-ecosystems", including those of the overseas territories, and to identify the main difficulties.

The following examples refer to types of ecosystems taken as a whole, for which the objective is to evaluate the services associated with their general biodiversity. Taking account of special contexts (very preserved character of the ecosystem, presence of species and/or remarkable habitats, etc.), some concrete ecosystems corresponding to these types may represent "remarkable biodiversity" e.g. forested areas or wet zones listed in the ZNIEFF inventory or designated in Natura 2000, and should be treated as such.

---

[1] Which, according to our calculations, values the biodiversity at less than 1 dollar per hectare per year.

## 5.1. Methodological options

### *a. The services considered*

In this study of various services, the workgroup opted to **evaluate the annual value of a service compared to a situation where it would be completely cancelled if the ecosystem in question were destroyed.** The fact that it is replaced by an ecosystem still providing a part of this service, or by a non-ecological service providing a similar function (for example the geological storage of $CO_2$ or the chemical production of oxygen) will be considered in the cost/advantages report for the different possible options, but a positive accounting seems to us to be preferable for an ecosystem, given the "potential costs avoided" that are associated with its conservation. Let us remember in fact that the socio-economic calculation is to have as input estimates for the advantages that are as pertinent as possible and not, as we will discuss later, to fix prices for a possible remuneration of these services.

Consistent with this option, the **group has decided to consider not only "dynamic" services, that is, fluxes** (carbon sequestration, water production, tourism, etc.) **but also "static" services** (stability of the soil, conservation of carbon stocks), considering that the value of the potential loss of these services in the event of destruction of the ecosystem (increase in erosion, liberation more or less rapid of $CO_2$) should, due to the fact that it is to be avoided, be credited to these ecosystems.

### *b. The total updated value of the services*

To standardise the estimates, **all the services have been expressed in monetary units per hectare per year.** Besides the question discussed in VII.3.4 of the limits of a standardization by unit of surface area, the question of the calculation of a total value of the service has also to be asked. The general conclusions of the literature of chapter V encourages us to follow the recommendations of the report of the Lebègue commission on discounting, while drawing attention to the hypotheses for changes in the relative price.

The idea that the prices of the ecosystem services increase compared to those of manufactured goods and services is certainly reasonable, and we have suggested an annual increase of 1% as a guideline. This value would be consistent with the order of magnitude of the gain in productivity in the economy[1]. We have described some situations where losses would be worrying and may lead to more rapid appreciations.

We should however notice that the **rhythm of revaluation will certainly not be the same for all services, nor necessarily regular over time.** It appears as a function of three types of variable:

- the effective dynamic of biodiversity and the ecosystem services, depending not only on the project being evaluated but also the choices

[1] The gains in productivity for industry have generally been between 1 % and 2% per year in the industrialised countries in the last few decades; but it is appreciably lower in the services sector. The productivity of the natural resources calculated by INSEE for the EU-15 varies around 1.3% per year between 1995 and 2004.

made on subjects that are different or elsewhere, and that may have an impact on this dynamic;

- the character more or less substitutable of the affected services, remembering that the substitution may operate at several levels (to offer the same service in another way or to offer different services that compensate the loss in well-being) and even have the effect of making the ecosystem services less precious;
- the hypotheses for changes in preferences for each service, whether it relates to a local request, like numerous recreational services that depend on population density, or its characteristics in terms of use of the spaces in question, or a wider request, like the production of commercialised goods or tourist requirements.

As in the case of carbon, simpler because of the existence of a single indicator, this dynamic of the request has to be modelled appropriately so as to be able to justify, case by case, the differentiated changes in the values.

**If the calculations are projected, on the simple basis of an annual revaluation of 1% and discounting of 4% over the first 30 years, then decreasing in line with the recommendation of the Lebègue report, we get a discounted total value equal to the value of the services of year zero (the moment where the evaluated changes take place) multiplied by 41.5.** Given the imprecision with which the values are known, this factor is not significantly different from 40. Supposing a more limited annual revaluation of 0.5%, this factor would be 33; for an accelerated rise in price of 1.5%, it will be necessary to multiply by 60.

The question of the function of carbon storage is specific because, in the event of conversion of the soils, the carbon sequestration function stops *a priori* and the non-sequestered units have to be evaluated according to their shadow value, this figure for France coming from the report of the Quinet commission. The carbon storage function is interrupted as well, but this is in the form of a massive de-stocking at the moment of conversion (even if a part of the wood is used in a sustainable manner). The simplest is therefore to value the stock thus released at its shadow value at the moment when this occurs.

## 5.2. The coral reefs

We will start with this example to illustrate an extreme case of an emblematic tropical environment and show the values that the ecosystem services may be given in this case. We will also be able to challenge the idea that this is an “ordinary” biodiversity study. We should remember that these coral reefs are environments often associated with biodiversity “hot spots”, that they are greatly threatened by almost all facets of global change (pollution, global warming and rising seas, demographic increases, introduction of species, etc.) and that France, hosting in the whole of its inter-tropical territories 10% of the planet’s reefs, has a particularly heavy responsibility in this area.

The data that we present comes from a recent bibliographic compilation of the MEEDAT (D4E, 2008), to be contacted if more details are required. Table VII-3 summarises the range of values for the various assessments made, the data being expressed per hectare of coral reef. It show, that, even if the extraction services have significant values (fishing in particular contributes to the feeding of populations that depend to a great extent on this resource), these extraction services only constitute a small part of the total economic value of these environments. Two items appear to be particularly important:

- the function of coastal protection, directly (breaker role of the reefs) or indirectly, facilitating the installation of coastal mangrove forests;
- tourism value, often contributing greatly to the gross domestic product of a number of island states.

Table VII-3: Estimates given in the literature
for the ecosystem services of the coral reefs
*(in 2007 dollars per hectare)*

| Services | Value proposed |
|---|---|
| Gathering services | |
| - Fishing | 40 – 900 $ |
| - Fish keeping | 1 – 10 $ |
| - Aquaculture | 1 – 65 $ |
| - Construction materials | 10 – 270 $ |
| Regulatory services | |
| - Coastal protection | 8 – 40,000 $ |
| - Biodiversity | 1 – 660 $ |
| - Water treatment | 75 – 100 $ |
| Cultural services | |
| - Tourism and recreational services | 100 – 37,000 $ |
| - Research and education | 5 – 200 $ |
| - Value of existence and option | 3 – 160 $ |
| TOTAL | **1,000 – 17,700 $** |

*Source: MEEDAT (2008)*

Moreover, as shown in table VII-4, these items are subject to great variation, associated in particular with the density of the developments. This density influences both the value of the protection services and the volume of tourism.

Table VII-4: Estimation of the economic value of the coastal protection services
and of the recreational services (tourism) of the coral reefs

| Site | Coastal protection | Recreational value | Reference cited* |
|---|---|---|---|
| Global | 4,500 | 5,000 | Costanza, 1997 |
| Indonesia | 250 – 7,000 | 83 – 13,400 | Cesar, 1996 |
| Philippines | 100 – 2,000 | - | White, 2000 |
| South-East Asia | 80 – 2,000 | 325 – 3,700 | Burke, 2002 |
| Sri Lanka | 3 500 – 12,000 | - | Cesar, 2003 |
| French Polynesia | 1,180 | 10,320 | Charles, 2005 |
| Northern Marianas | 1,860 | - | Van Beukering, 2006 |
| Galapagos | - | 100 | De Groot, 1992 |
| Thailand | - | 7,300 | Seenprochawong, 2003 |
| Caribbean (St. Lucia) | 250 | - | WRI, 2006 |
| Caribbean (St. Johns) | - | 37,000 | Posner, 1981 |
| Caribbean | - | 2,000 | Chong, 2003 |

* See the MEEDAT report for the complete bibliographic references for these works

*Source: MEEDAT, 2008, data in 2007 dollars/ha/yr*

**In the case of the French coral reefs, often situated in popular tourist areas (Polynesia, Caribbean, New-Caledonia), it seems legitimate to opt for a value at the high end of the range, of the order of 5,000 to 10,000 €/ha/yr**, coming to a total economic value of several billion euros per year. In comparison, the 2.8 million euros raised since 2000 in the IFRECOR action plan for a sustainable management of the coral reefs appear more than justified.

In respect of the great uncertainties about the value of the services, the example that follows will help us to identify some of the factors responsible.

## 5.3. The wet zones

We have available three meta-analyses of values for wet zones (WZ), the ecological services that they procure and their biodiversity: Brouwer *et al.* (1999), Woodward and Wui (2001) and particularly Brander *et al.* (2006), who are interested really in making explicit the empirical values and analysing their variation factors.

These analyses vary with the geographical domain (temperate, tropical), the range of services of the WZ considered, and finally the method used (contingent assessment and, increasingly, the method of contingent choices). The aim in general is to give a standardised value e.g. in euros per hectare per year, which fits with the objective of this survey. But it may also be to measure the contribution that diverse variables make to the constitution of this average value. Which means that, at least in principle, an extrapolation can be made for any analogous project without having to start the analysis again. The development of the choice modelling methods means finer measurements of the variation factors of the value can be taken, taking account particularly of the heterogeneity of the preferences, and the interaction between the ecological characteristics of the WZ and how they are managed.

The highly changeable and, in some respects, broadly experimental character of the assessment methods causes great difficulty in analysing the results. Furthermore, the meta-analyses available introduce selection bias as they depend on the distribution of the base analyses, which do not correspond with those of the WZ. It may be possible to perform several analyses on the same WZ, at different moments or for different objectives. It is imperative to take into account this fluid aspect before any use in a specific decision procedure.

Finally, we should take note of the fact that the studies are made with various different methodologies, which means the calculation of the values per hectare may not always be easy. Thus, for example, the results of the contingent assessments or the contingent choices are expressed as individual WTPs. To convert them into values per hectare, a reference population needs to be defined, the corresponding aggregated value calculated and divided by the size of the WZ (which in fact is not always given in the articles).

The analysis of Brander *et al.* (2006), a meta-analysis of 190 studies resulting in the determination of 215 values using the whole range of methods available (91 of which being based on the market price), is the richest, but needs to be read with care. It results in an average value for WZs of 2,800 $USD_{1995}$ per hectare per year, but a median value of only 150 $/ha/yr; that is, in 2008 euros, a median of 170 euros and an arithmetical average of more than 3,000 euros. This discrepancy is unusually large and shows how asymmetric the distribution of the values is (the sample has no particular representative significance). It would be legitimate to think that sites of special interest are over-represented in the sample, which would mean that the median is a more reliable indicator of the spontaneous perception that we would have of the value of a hectare of "ordinary" WZ, having no particularly extraordinary element.

The values of European WZs (23 values, none in France) are shown to be, unsurprisingly (high GDP, high densities, very anthropised environments), the highest in the world (average around 10,000 €/ha/yr, but the median remains less than 200 euros). On average, for the 13 studies (of 215) that specified it, it is the biodiversity that obtains the highest average value (19,000 €/ha/yr) but a median value of barely 15 euros, ahead of the quality of the water and amenities. The use of these environments for provisioning in firewood or raw materials is at a lower average (100 and 350 euros), but with more "reasonable" medians (table VII-5). It should be noted that these studies do not include the question of greenhouse gases (sequestration or emission of $CO_2$, methane, nitrous oxides, etc.) while the wet zones certainly play an important role in this.

**Table VII-5: Average and median value (in $/ha/yr) of ecological services of wet zones**
*(The values are approximate being taken from a graph)*

| Service | Number of studies | Median | Average |
|---|---|---|---|
| Biodiversity | 13 | 13 | 16,500 |
| Amenity value | 48 | 50 | 6,900 |
| Wood for heating | 18 | 18 | 70 |
| Materials | 32 | 32 | 270 |
| Recreational fishing | 36 | 36 | 2,100 |
| Recreational hunting | 50 | 50 | 1,420 |
| Habitats and nurseries | 67 | 68 | 1,820 |
| Water quality | 25 | 28 | 7,300 |
| Water supply | 18 | 18 | 1,300 |
| Protection against floods | 26 | 25 | 3,900 |

*Source: according to Brander* et al.*, 2006*

We have in metropolitan France some studies based essentially on the WTP required for aquatic environments to attain a good ecological state. Thus, a study conducted amongst persons living around a 70 km section of the Loir (Sarthe) indicates that getting to the required state would have a value of 760,000 to 1 million euros per year, that is, between 11,000 and 14,000 €/km (D4E, 2006), this value including the natural heritage value for non-users of the water courses (between 186,000 and 287,000 euros per year). A similar study, already mentioned (see V.4), on a 25 km section of the Gardon downstream (D4E, 2007c), including the natural heritage value for the non-users of the watercourses, gives a value that is appreciably higher (2.86 million euros per year, that is, 110,000 €/km x yr). For these two studies, the consents to pay formulated by the persons interviewed are relatively close (between 15 and 35 €/household/yr) but the size of the population situated within the perimeter of influence of each of the rivers is only 28,000 households for the Loir while it is 100,000 for the Gardon, which explains the deviations in the aggregated benefits.

**In selecting for example an average "recreational" width of 20 m (including the banks) for these water courses, we would then have a WTP of the order of 5,000 to 50,000 €/ha/yr.**

The ministry for Ecology has also carried out three studies on wet zones, aiming to measure the willingness to pay of the neighbouring populations to contribute to their preservation. Two studies using the contingent assessment method were conducted on wet zones that are very heavily used for recreational purposes and bird-watching: the one in 2003 on the lac du Der, in Champagne-Ardenne and the other in 2005 on the estuary of the Orne, near Caen. A third study, in 2005, used the choice experiment method on the Natura 2000 site of the marshes of the Erdre, near Nantes. Different protection programmes for the site have been suggested to the residents interviewed, and this was used to measure their preferences with greater precision, the results showing them to be much more sensitive to the protection of endangered species and the maintenance of the river banks than the maintenance of the canals and meadows. More recently, Beaumais *et al.* (2008) have been interested in the wet zones of the Seine estuary, subject to heavy pressures in terms of urbanisation, extension of the ports and industrialization. The results of these four studies are summarised in table VII-6.

**Table VII-6: Estimation of residents' willingness to pay for four wet zones**

| Environment | Method used | WTP /yr x household | Size of the population in question | Surface area of the WZ studied | WTP/ha/yr |
|---|---|---|---|---|---|
| Lac du Der | Contingent assessment method | 30-33 $€_{2003}$ (average WTP) | 117,000 in habitants in the communities close to the lac du Der, around 46,600 households | 4,800 ha (surface area of the body of water) | 291-320 $€_{2003}$ |
| Estuary of the Orne | Contingent assessment method | 30-66 $€_{2003}$ (average WTP) | 13,500 inhabitants of the 5 communities situated at less than 20 km from the site, around 5,400 households | 900 ha | 179-394 $€_{2003}$ |
| Marshes of the Erdre | Choice experiment method | 34 $€_{2005}$ (average WTP to move from averagely ambitious to ambitious protection) | 22,555 households in the 7 neighbouring communities of the marshes of the Erdre | 2,500 ha | 307 $€_{2005}$ (average WTP to move from averagely ambitious to ambitious protection) |
| Estuary of the Seine | Contingent assessment method | 18-46 $€_{2008}$ (average WTP) 14-44 $€_{2008}$ (median WTP) | 1,17 million persons living on the Seine estuary, in around 500,000 households | 14,000 ha | 659-1 652 $€_{2008}$ (average WTP) 517-1 563 $€_{2008}$ (median WTP) |

*Source: CASE, Biodiversity group*

**Again, while the amounts for the expressed willingness to pay are relatively close in the four studies, around 30 euros, this example shows up to what point the**

**definition of the population in question, that is, the perimeter of influence of a natural space, has an impact on the estimation of a consent to pay per hectare.**

## 5.4. The case of the services of temperate forests

As we have indicated, temperate forests seem, in terms of ecosystem services, to be at an intermediate level between the extremes representing, at the bottom of the scale, the environments devoted to intensive monocultures and, at the top, the tropical environments of great biodiversity, such as the mangrove forests. Although less studied than tropical forests, they have been the subject of quite a lot of work that often, unfortunately, doesn't distinguish forests that are Mediterranean, temperate and boreal. We refer the reader in particular to the compilations of Krieger (2001), Pearce (2001), Turner *et al.* (2003), Brahic and Terreaux (awaiting publication) and the recent report of Mullan and Kontoleon (2008) used by the EU for the interim report already mentioned. In this case we are going to be able to discuss the uncertainties in estimating the value of the various services and the source of these uncertainties. This is why we propose applying the previous analyses to the case of the French temperate forests and for which we have studies covering several services (MAP, 2006; IFEN, 2005; FPF, 2008; MEEDAT, 2008), this to see up to what point it is possible of go in the estimation of reference values for such ecosystems using a cost/advantages approach.

A similar study could be made for the tropical forests of the overseas departments and territories, based on a critical analysis of the significant amount of work that has been published.

**These "elementary" reference values, based on the current state knowledge, constitute, in our view, the "point of input" for a procedure to set global reference values, a procedure that we will consider later.**

### *a. Global analysis*

The table VII-7 summarises a number of global analyses strictly distinguishing the value of the wood (and forage in the case of grazed forests) and the other values associated with these ecosystems.

Table VII-7: Value of the ecosystem services
(in dollars, euros or pounds sterling per hectare per year)
for different Mediterranean, temperate and boreal forests

| Country | Type | Value of wood | Others | References |
|---|---|---|---|---|
| 1. Turkey | Medit. | Not indicated. | 61,5 $ | OCDE, 2001 |
| 2. Turkey | Medit. | 33 $ | 6 $ | MEA, 2005* |
| 3. Syria | Medit. | 2 $ | 86 $ | *Id.* |
| 4. Croatia | Medit. | 131 $ | 118 $ | *Id.* |
| 5. Italy | Medit. | 88 $ | 161 $ | *Id.* |
| 6. Tunisia | Medit. | 82 $ | 59 $ | *Id.* |
| 7. Algeria | Medit. | 20 $ | 20 $ | *Id.* |
| 8. Portugal | Medit. | 147 $ | 186 $ | *Id.* |
| 9. Morocco | Medit. | 49 $ | 19 $ | *Id.* |

| | | | | |
|---|---|---|---|---|
| 10. France | Medit. | 25 € | 215 € | MEEDAT, 2008 |
| 11. Scandinavia | Boreal | 45-85 $ | 35-50 $ | Turner *et al.*, 2003 |
| 12. Canada | Boreal | 49 $ | 41 $ | Anielski and Wilson, 2005 |
| 13. Not detailed | Temp./Bor. | 25 $ | 277 $ | Costanza *et al.*, 1997 |
| 14. Switzerland | Temperate | 218 € | 1,867-3,846 € | Rauch, 1994; Alfter, 1998 |
| 15. Great-Britain | Temperate | | 349 £ | Willis *et al.*, 2003 |

* N.B.: for the studies presented by the MEA, the estimates are taken from the graph as we have not found the exact numerical data.

*Source: CASE, Biodiversity group*

As far as the value of the wood is concerned, the documents do not always indicate if this is the value of standing timber (strictly representing the value of the ecosystem service) or the market value after processing. We will return later to this point. We have included in this table only the studies that explicitly sum the various services. Later, we will use other work for particular services.

The dispersion of these values is quite marked, particularly because the studies are exhaustive to differing degrees for the ecosystem services studied. However we frequently see that the values other than the direct productions of wood and forage are close, and in fact quite clearly higher than those associated with these productions.

We will now refine the analysis, focusing on the case of the French forests and using the typology of the services proposed by the *Millennium Ecosystem Assessment*; that is, the regulatory and cultural services. We will also find in Stenger *et al.* (2009) a description, with remarks, of these services adapted to forest ecosystems. **We are going to indicate also the data available on the role of biodiversity in the modulation of each of these services.**

### *b. Extraction services*

*The production of wood*

In 2007, French forests covered 15 million hectares and 59 million $m^3$ was collected (22 million of which being consumed directly). This represents an average harvest of 4 $m^3$/ha/yr (FPF, 2008). This value of 3 to 4 $m^3$/ha/yr may be considered to be quite stable (Lebreton and Valleuri, 2004; IFEN, 2005) but we consider today that it represents only a part of the annual increase of the volume of standing timber, estimated at between 103[1] and 135 million[2] $m^3$, depending on the method used. A higher rate of processing would therefore be possible and is in fact envisaged, but it comes up against numerous difficulties, technical, economic (accessibility and costs of processing and transaction) and social (sentiment of the owners, land ownership structures) and we will therefore use this value of 4 $m^3$/ha/yr. This average value is based on a wide range (e.g. the forest of Les Landes may produce as much as 10 $m^3$/ha/yr) and should therefore be adapted to the different regional contexts.

[1] National forest inventory, that measures the growth in volume of the main trunk, from the stump to a diameter of 7.5 cm.
[2] The AFOCEL 2007 study taking into consideration all the ligneous aerial biomass that may be harvested, including secondary branches.

In financial terms, the price varies a great deal depending on the tree species (more than 100 €/m$^3$ for oak and less than 30 €/m$^3$ for pinasters). MAP (2006) gives an annual value after exploitation of 1.68 billion euros for wood marketed in the period 1998-2002, of volume 38 million m$^3$; that is, an average of 44.2 €/m$^3$ (55.8 €/m$^3$ for constructional timber, 20.8 €/m$^3$ for industrial timber, 32.3 €/m$^3$ for firewood). If we value the 22 million m$^3$ consumed off-market at the price of the marketed wood, **we arrive at a total value of production of 2.4 billion euros, 160 €/ha at 2002 prices.**

However, these values include the costs of processing, that are estimated at 20 to 25 €/m$^3$ (Michel Badré, personal communication) and it would be more appropriate to assess the ecological service in the strict sense by using the value of the standing timber. Montagné and Niedzwiedz (2007) give, for 2003, an average standing timber price of 24.9 €/m$^3$ for marketed wood and estimate at 8.2 €/m$^3$ the value of the non-marketed wood. **Applying these values, the total production falls to 1.13 billion euros, that is 75 €/ha.**

**We shall use this last estimate, but it seems desirable to also mention the value after processing, wishing to ensure a homogeneity both with respect to the other forest extraction services, estimated, as we will see, at the market price (and so including collection costs), and also with the extraction services of other ecosystems such as fishing, for which the concept of "standing wood price" has no equivalent.** It remains a methodological point that is rarely mentioned and needs to be clarified.

To optimise this service, it could be thought that the choice of a well-performing tree species is preferable to the use of a diversified stand (population) and that biodiversity plays only a negligible, or even negative, role. But this view needs correcting with two remarks: firstly, **the plantation of species of tree suited to the various current (and, above all, future) pedo-climatic situations (species or varieties of species) supposes for this very reason that we base ourselves on the existing biodiversity and know its characteristics**. So, during the winter of 1985, 100,000 hectares of pines of Portuguese origin froze in the Les Landes region: this was doubtless a high-yield population, but more sensitive to the cold than the local population and the producers hadn't anticipated such extreme conditions. Similarly, the administrative forest seed dry kiln of La Chaise-Dieu, after the last war, sold seeds of "noble Auvergne pine", the cones of which had been bought from shepherds in les Causses, where the collection was effectively much easier than from the high branches of the crystalline massifs. The plantations of Scots pine still sometimes bear the mark (Yves Poss, personal communication). Secondly, the **superiority of a homogeneous stand is only true in the short term and in a stable environment** and this choice may turn out to be disastrous when unanticipated *events* occur. This reasoning relates to to attacks by crop-destroying insects, which may be more limited if there is a diversified bird population, itself associated with a tree population of a great diversity of species and ages. Thus, the study 12 gives an estimate of 18.2 USD/ha/yr (2002 value) as the beneficial effect of the predation by birds on the caterpillars devastating conifers in the United-States. We also find in Jactel and Brockerhoff (2007) a meta-analysis of a hundred publications relating to this protective effect of biodiversity.

*Gathered forest products*

Diverse gathered products come into this category (fruits, flowers, mushrooms, dead wood) and also hunting. These products may or may not be commercialised, hence the difficulty of precise estimations. Depending on the case, these services are estimated separately, globally or incorporated into the idea of recreational value to avoid the risk of double accounting. Table VII-8 gives some estimates from the literature.

**The values obtained appear to be relatively consistent, even if there are clearly strong local variations, attested by a number of conflicts of use, (particularly for hunting and mushrooming).**

It should be noted that for mushrooms, the estimate only relates to truffles, boletus and chanterelle mushrooms and only the quantities sold (3,000 to 4,000 tonnes depending on the year). A telephone survey in 2002 (MAP, 2006) assessed the harvest for immediate consumption at 12,650 tonnes. Added to this were 4,360 tonnes of fruit (chestnuts 80%) and 330 tonnes of flowers and other decorative elements.

In terms of hunting, the values may vary considerably depending on the activities included. So, if hunting is considered as a supply service, then only the commercial value of hunted game is estimated. MAP (2006) gives an estimate of 61 million euros for the large game alone [1], hence a figure of 4 €/ha/year. **But it seems to us to be more appropriate, in the French context, to consider hunting as a sporting and recreational activity and assess it later under cultural services.** It is because game is wild (or "set up" as in the case of repopulation) that it is the hunter rather than the buyer on the market who accepts specific costs to procure it. So it really is the creation of economic value induced by the ecosystem services. The question is different[2] for supply services such as wood, for which the users only consent to the costs strictly necessary for its acquisition, and are ready to use other materials depending on relative prices.

**Table VII-8: Estimate of the value of the harvesting of non ligneous products (in dollars or euros per hectare per year)**

| Country | Denomination | Value | Reference |
|---|---|---|---|
| Turkey | Non ligneous forest products | $ 18.4 | 1 |
| Canada | Non ligneous forest products | $ 2.4 | 12 |
| Scandinavia | Gathered products | $ 10.15 | 11 |
| Switzerland | Gathered vegetable products<br>Hunting | 22 €<br>5 to 7 € | 14 |
| France (medit.) | Non ligneous forest products<br>Hunting | 7 €<br>5.8 € | 10 |
| France (global) | Vegetable products<br>Hunting | 7.6 €<br>5.8 € | IFEN, 2005<br>*Id.* |

[1] Deer, roe-deer and boar, that is a total of around 23,000 tonnes in 2002-2003, including for immediate consumption, with an average commercial value estimated globally at 2.64 €/kg.
[2] The analogy would be possible if there existed amateur lumberjacks, spending considerable amounts of money to practice their favourite sport!

| | | | |
|---|---|---|---|
| | Secondary products (truffles, hunting, etc.) | 7.3 € | Montagné *et al.*, 2007 |
| | Forest honey | 1.3 to 2 | MAP, 2006 |
| | Mushrooms* | 0.7-1.3 | *Id.* |
| | Products from hunting | 4 € | *Id.* |
| | Rent for hunting (private forests)** | 19 € | FPF, 2008 |
| | Rent for hunting (state forests)*** | 17 € | MAP, 2006 |

* This figure only relates to the quantities put up for sale.
** This study is of private forest, only 11% of which is rented for hunting. The figure obtained (24 million euros for 1.28 million hectares rented) cannot be extended to the whole of the French forests without precautions).
*** 31.4 million euros for a total surface area of 1.82 million hectares of state forest. The study does not say if the whole surface area may be rented.

*Source: CASE, Biodiversity group*

**As a first approximation, a reference value for France of 10 to 15 €/ha/year does seem feasible for these gathering services (except hunting but including immediate consumption of the other forest products) .**

The role of biodiversity in the production of these services is quite clear, since it determines the very diversity of the products sought.

### *c. Regulatory services*

We shall examine more particularly the aspects linked to carbon storage, the water cycle, the protection and preservation functions of the habitats and biodiversity.

*Carbon storage*

Table VII-9 gives the results of some of the available studies. These studies deal essentially with $CO_2$ absorption (we will say something later about the role of forests in the modulation of the atmospheric content of other gases). **The values appear relatively dispersed, but it is advisable to distinguish the biological and economic dimensions of the phenomenon:**

- at a biological level, the net sequestration of carbon in temperate and boreal forests of the Northern hemisphere has been the subject of numerous studies, in particular the recent work of Magnani *et al.* (2007). It can vary between – 3 to + 6 tonnes of carbon per hectare per year depending on the age of the forest, the latitude (with a maximum for the French latitudes, see table) the forest species and the volume of atmospheric nitrogen. For French forests, it is generally held to be an **average of a tonne of carbon (3.6 tonnes of $CO_2$) per hectare per year**, with a tendency to increase due to an under-exploitation of standing timber and excessive growth of the trees in the course of the last decades, in which atmospheric nitrogen without a doubt plays a large part (Dupouey *et al.*, 2002; IFEN, 2005; ONF, 2006). Thus, the last value declared by France under the Kyoto protocol was 66.87 million tonnes of $CO_2$ in 2006 for the whole of the metropolitan forests[1]; that is, 1.2 tonnes of carbon per hectare;

[1] Excepting extension of the forest surface area

Moreover, **it is advisable to bear in mind not only the annual sequestration but also the long term storage function.** An important part of the carbon absorbed is not immediately remobilised (as would be the case for example for biofuels) but stored either in the aerial part, or in the subterranean part of the forest, much more developed in temperate forests than in the tropical forests. We estimate this stock at around 150 t/ha (MAP, 2006), about two thirds of which is underground[1]. As indicated in the methodological options (see VII.5.1), it is therefore legitimate to give a value to this stored capital[2], considering that this is a protection function contributing to the slowing down of the greenhouse effect. It is difficult to estimate precisely the fraction of this capital stored for the very long term (at least 30 years). It is without doubt the case for the major part of underground carbon (in the absence of deforestation) and a significant part of aerial carbon, at least 20% of which remains immobilised after collection as solid wood, veneer or for chip board[3] (FPF, 2008). **By taking 25% and 75% as the respective rates of long term immobilisation of stocks of aerial and underground carbon, we arrive at a stock with long term value of around 90 t/ha;**

**Table VII-9: Value of forest carbon sequestration, soil included (in dollars or euros per hectare and per year)**

| Study | Value | Reference |
|---|---|---|
| Turkey | 26 $ | 1 |
| France (medit.) | 24 € | 10 |
| France (global) | 22 to 150 € | IFEN, 2005 |
| Europe latitude 35-45 | 29 $ | EU, 2008 |
| Europe latitude 45-55 | 51 $ | *Id.* |
| Europe latitude 55-65 | 19 $ | *Id.* |
| Europe latitude 65-71 | 10 $ | *Id.* |
| Canada | 6.1 $ | 12 |
| Scandinavia | 10 to 15 € | 11 |
| United-States | 44 $ | Dunkiel and Sugarman (1998, *in* Krieger, 2001) |
| United-States | 29 $ | Loomis and Richardson (2000, *in* Krieger, 2001) |
| United-States | 28 $ | Pimentel *et al.* (1997, *in* Krieger, 2001) |

*Source: CASE, Biodiversity group*

- on an economic level, the tonne of $CO_2$ is valued in diverse ways that are not always described. The Canadian study suggests 10.1 dollars, studies in the United-States opt for 18 dollars, the British study (15) gives extremely low

[1] Dupouey *et al.* (2002) give the following distribution: 32% for large round wood and branches, 2% for the leaves, 2% for the under-storey (shrubs) and dead wood, 6% for the litter, 7% for the roots and 51% the soil.
[2] This is indeed an ecological service, in so far as this continuous storage depends on the biological activity of the forest, which is also the case for both aerial and underground carbon, rapidly remobilised when the forest is cut. The reasoning is of course not valid for fossil carbon (petroleum and carbon and also limestone).
[3] We are not going to go into here the complex debate between those arguing for *in situ* storage, through an extension of the processing cycles, and the defenders of *ex situ* storage, with a greater use of wood materials for long term uses, in particular because *ex situ* storage, while it can be considered a service, does not constitute an ecological service.

values (£1.8 to 3.6 [1]) and the French IFEN study explores a range starting at the lowest recorded on the quotas market (6 euros) to a maximum of 40 euros. **As we have decided to use a cost/advantages approach, we should take as reference the estimate of the damage associated with a tonne of $CO_2$ emitted into the atmosphere.** As we have indicated previously, the Stern report (see CASE, 2008) proposes three values, depending on the policy adopted to stabilise the atmospheric concentration of $CO_2$: 85 dollars in the absence of any policy of limitation, 35 dollars for a stabilisation at 550 ppm, 25 dollars for a stabilisation at 450 ppm. Following this study, the Department for Environment, Food and Rural Affairs (DEFRA) has proposed to the British government a value of £26 per tonne.

We have seen however that this estimate based on a cost/advantages has been the subject of a number of criticisms (see VII.1.4), leading CASE to suggest a cost/efficiency approach (Quinet, 2008), resulting in a shadow value of 100 $€_{2008}$ in 2030, with a starting estimate of 32 euros for 2008. Aside from the pertinence of the previous criticisms, we think it is preferable to choose this value, if only to ensure consistency with the other parts of the socio-economic calculation (estimate of the pollution) that make use of this reference value.

Finally, an approach based on the cost of replacing the service could be envisaged, considering the long term cost of $CO_2$ sequestration by other methods, in particular deep geological storage. Values of 40 to 70 €/t can be found in the literature[2], rather higher than the reference value that we propose, but both the technical and economic aspects of these methods of storage are changing fast and these values are therefore to be viewed with caution.

We will see that all these approaches involve a large increase in these values, quite possibly reaching between 50 to 100 €/t by 2030 and something between 150 to 350 €/t in 2050.

With this value of 32 €/t, the sequestration function would be valued in 2008 at 115 €/ha/yr.

For the storage function, the valuation depends mainly on the rate of remuneration chosen for this fixed capital. As it is the price accorded to slow down a given damage that is to be estimated, and so to estimate a preference for the present, the group proposes to take an annual remuneration rate, short and medium term, that is identical to the rate of discounting accepted today, 4%. With this hypothesis we get a value of 414 €/ha/yr, that is, 529 €/ha/yr for the functions of sequestration and storage accumulated together, with a perspective of 800 to 1600 euros in 2030. If we wish to apply a lower weighting to this protection function, say, a remuneration rate of only 2% – which seems to us to be a minimum – we get a value of 322 €/ha/yr for this service.

Another approach for this rate could be based on the speed of mineralization of the carbon after the elimination of permanent vegetal cover. Soussana *et al.* (2004) suggest a value of 0.95 t/ha/yr for 20 years for the carbon in grassland soils, about 2%

---

[1] There is perhaps in this study a confusion between a tonne of carbon and a tonne of $CO_2$.
[2] See, for example, the site of the Ministry for Industry.

per year. If we assume a higher value for aerial carbon, we re-enter the range of 2% to 4%.

For the effect of biodiversity on this carbon sequestration, we refer the reader back to the preceding reflections on the production of wood. Mention of the beneficial effect of biodiversity is to be found in particular in the analysis of Bolker *et al.* (1995) who models the climate warming effect in accordance with whether the stand is mono-specific or made up of a number of species and so likely to see a gradual modification in its composition over the long term (50 to 150 years). This capacity of adaptation of composite stands[1] leads finally to a gain of the order of 30%, in terms of carbon sequestration capacity, compared to a mono-specific stand.

*Other atmospheric gases*

Besides $CO_2$, the ecosystems play a major role in the regulation of the quasi-totality of the gases of the atmosphere (oxygen and ozone, nitrogen in its different forms, methane, water vapour, etc.), and in particular gases like methane, with a capacity to contribute, for a given quantity, much more than $CO_2$ to the greenhouse effect. However, before envisaging an economic estimate, the quantitative balance sheets and their sources need to be established. Thus, in the case of methane, the role of the ruminants and paddy fields is often mentioned but other sources are possible, like certain invertebrates having methanogenic digestive fermentations (termites, molluscs) or, maybe, vegetables in certain conditions (see the recent debate started by the article by Keppler *et al.,* 2006, identifying methane emissions from the leaves of terrestrial plants).

An interesting case is that of oxygen, as **it illustrates the large divergence that is possible between two estimating methods, in this case a "production function" approach and a replacement costs approach.**

In effect, any sequestration of a molecule of $CO_2$ is accompanied by the dissociation by photolysis of a molecule of water, and so the liberation of a molecule of oxygen. This leads to the production of 2.7 tonnes of oxygen per tonne of sequestrated carbon.

A production function approach leads to an estimate of the consequences of a reduction of this photosynthetic activity on the oxygen content of the atmosphere, and the possible economic consequences of a possible reduction in this content. **In this case, we can legitimately ignore this function.** Effectively we reckon that, even under the extreme hypothesis of a complete stop of photosynthesis, leading in about twenty years to an oxidation of virtually all surface organic carbon (including that of our own species), the atmospheric oxygen content only drops 1% and about another 4 million years would be needed for all this oxygen to be exhausted by the oxidation of the metals in rocks [2] (Rasool, 1993).

---

[1] Population with stands of mono-specific origin may also gradually be enhanced, as is demonstrated by the evolution of forestation of Austrian pine, used for restoring mountain terrain (Y. Poss, personal communication).
[2] This inertia is due to the fact that atmospheric oxygen has accumulated above all due to the immobilisation of the $CO_2$ in carbonated rocks and that the this storage is stable in the very long term.

A substitution costs approach would however, with current techniques, give higher values. To obtain the equivalent of the photolysis of water, we could envisage electrolysis or high-temperature thermolysis[1]. In the case of electrolysis, by limiting ourselves to only the energetic cost of dissociating the water molecule (282 kJ/mole), we estimate that for every kg of oxygen produced the minimum quantity of electricity required is 5 kWh (source: website of the French Chemistry Society); that is, 13,500 kWh for the 2.7 tonnes produced by a hectare of forest. **Even at the current cost of French electricity, about 0.05 €/kWh, the cost of this substitution service would amount to 675 €/ha/yr. This calculation only gives us an idea and is very theoretical, as, if this activity were to be really developed, it would be possible to profit greatly from the co-product, hydrogen, and use cheap period kWh. But it does help us to think about other aspects and not just the carbon cycle.**

*The water cycle*

In addition to carbon sequestration, an important, often mentioned function of forests is their role in regulating the water cycle and the production of quality water.

In quantitative terms, Krieger (2001) cites a study by the United States Forest Service which estimates that national forests, representing an area of 76.9 million hectares[2], "produce" approximately 8,600 $m^3$ per hectare per year of high quality water, which could be valued at \$41 per 1,000 $m^3$ if it was entirely consumed. This would therefore represent a service of the order of \$354 per hectare per year.

On further analysis he proposes distinguishing water consumed, or about 6% of this total, with a value of \$32 per 1,000 $m^3$, which would represent a service of about \$20 per hectare per year. But he believes that non-consumed water contributes to river flow and therefore to the river's function as a biodiversity host, tourist attraction (fishing, water sports) and for other uses (irrigation, hydroelectric production) and gives values ranging from \$0.8 to \$36.5 per 1,000 $m^3$ for these purposes (using methods such as contingent evaluation, displacement costs or marginal economic value of increased flow), thereby providing an additional value of between \$7 and \$390 per hectare per year.

As regards France, the "average effective production" of water in the territory is about 3,200 $m^3$ per hectare per year, or 36% of the average precipitation[3] (the rest is accounted for by evapotranspiration). We found no specific figure for the forests but we can assume a close figure, or even higher since most forests are found in regions of higher precipitation. **However, we propose not to adopt this approach for overall valuation and we refer the reader to specific studies. This proposal seems to us to be justified for several reasons:**

[1] The industrial production of oxygen usually involves less costly methods such as the distillation of liquefied air, but these methods don't really produce oxygen, they merely separate the constituents of the atmosphere.
[2] This study uses American units (*acre* for surface areas, *acre-foot* for volumes). We hope that we have made the conversions in units correctly!
[3] Average annual precipitation over 50 years is 889 mm, or 486 billion $m^3$ for the entire metropolitan territory.

- firstly, **in terms of annual water budget, strictly speaking, the water surplus from forests should only be valued in comparison with another land use.** This positive effect of the forest cover on local or regional precipitation, even if it is often mentioned and seems to be accepted in principle, at least for tropical rainforests[1], is still very controversial and very difficult to quantify. Some even consider that the higher evapotranspiration of forests would more than offset any positive effect on precipitation, leading to a lower water budget than for grasslands or cultivated regions (Willis *et al.,* 2003). In particular, the deeper roots of trees help to maintain strong summer evapotranspiration and may accentuate the magnitude and duration of minimal water flows. This means that a change in use of a territory only marginally changes the value of this function and it is therefore unnecessary to evaluate it precisely;
- **there is a risk that data on the regulation of the water cycle (flood management, low water support) may be included in the assessment of other functions**, including those that focus on the recreational value of forests and water projects[2] or on the protection function. We can however envisage specific assessments of this spreading effect on the hydrograph (flow distribution over the year). Thus, Guo *et al.* (2000) have calculated the surplus electricity production associated with this spread[3] for a dam on the Yangtze. The watershed is in this case 98% covered by forests and shrubland and the authors estimate 170 Kwh per hectare per year (or US $2.6) to be the positive effect of this cover[4]. Note that mixed forests would have the most favourable role, followed by coniferous forests and shrublands, and that these mixed forests would be about 15 times more efficient than lands cultivated for the purpose;
- finally, on a larger scale, **these services could also be attributed to the affected aquatic ecosystems**, thereby generating problems of double accounting. On this last point, it seems to be legitimate to continue to attribute this service to forests (implying the problem of double accounting is dealt with when accumulating the ecosystems), in so far as any degradation of forest ecosystems would lead effectively to a change in these services. It will be necessary therefore to attribute the cost to the source, that is, to those responsible for these degradations.

**However, if we do not propose to evaluate the quantitative dimension of water production, it seems legitimate to retain the qualitative aspect, namely the production of high quality water and the economies of water treatment that result.** Indeed, as rain falls on a forest, a highly cultivated region or an asphalted surface, the water which results will have significantly different chemical properties. In this field, the most famous example is New York City, which decided at the end of the nineties to invest 1.4 billion dollars to restore a 32,000 hectare watershed, rather than setting up a treatment plant costing 3 to 4 times more. Krieger (2001) cites two other

---

[1] More generally, we acknowledge the phenomenon for the large forest stands for which rainfall is essentially "endogenous", that is, fed by local evapotranspiration. When these rains are mainly exogenous and especially oceanic air masses (as in Western Europe), there is still active debate.
[2] For example, one effect of low water level support will be to encourage nautical sports and tourism.
[3] In the case of high water flow rates, not all water can pass through the turbines. Conversely, with very low water levels, a minimum flow must be maintained and the water cannot be stored.
[4] An effect that however only represents 0.3% of total electricity production of the dam.

American studies and Pointereau (2007) presents the case of the city of Munich, which early in twentieth century began a policy of controlling its watershed and agricultural and forestry practices. This city supplies 1.3 million inhabitants with potable water without prior treatment, with a nitrate content of 12 mg/l, which is decreasing (a single preventive chlorination in the last fifteen years). The price of water in 2007 is 2.74 €/m³ (including wastewater treatment), while the average in Germany was 5.09 €/m³.

Table VII-10 summarizes these studies. **We note that the resulting ecosystem service can often have considerable values when the population concerned is significant**, representing several hundred euros per hectare per year if replacement costs are considered. However, if we relate these values to the volume of water having reduced cost of treatment, the costs remain modest. For example, apparent consumption in New York City can be estimated as 3 billion m³ per year (1,000 litres per day per inhabitant x 8 million inhabitants), which gives about $20 per 1,000 m³.

Table VII-10: Watershed protection investments

| Study | Total annual expenditure* | Per hectare per year | Per m³ | Annual replacement cost* |
|---|---|---|---|---|
| New York | $ 56 million | $ 1,875 | $ 0.02 | $ 460-540 million |
| New Jersey | $ 2.2 million | $ 300 | Not specified | $ 6.4 million |
| Portland | $ 0.92 million | $ 35 | Not specified | $ 8 million |
| Munich | € 600,000 | € 150 | € 0.01 | Not specified |

* dividing total investment by 25.

A rough calculation can be made to estimate the value of this service for the entire territory of France: based on apparent daily consumption of about 300 litres per day per inhabitant[1] (or 100 m³ per year) and assuming that 30% of this volume comes from a forested area (forest covers 28% of the territory), it can be estimated that 110 m³ per hectare per year is the contribution of forests to the production of potable quality water. The cost of water treatment is a few eurocents per m³ for a simple mechanical filtration but can reach 0.50 Euros for complex techniques using activated carbon filters or ultrafiltration membranes (Corisco-Perez, 2006). **As a result, 0.40 euros in savings would lead to about 44 euros per hectare per year.**

**We propose doubling this figure and settle on a value of 90 Euros as a reference value, taking into account four factors which are difficult to quantify (more detailed studies will be done) but which significantly increase the role of forests:**

- the location of many forest stands in regions of high precipitation. Combining 1961-1990 precipitation data from the AURELHY model of Météo France and forest cover data from CORINE Land Cover, Badeau (personal communication)[2] obtained an average precipitation on French

[1] The domestic consumption figure of 150 litres is always mentioned, but these are household consumptions recorded "at the meter". In fact, the total volume of treated water is 17.9 million m³ per day, or 300 litres per person.
[2] We thank Vincent Badeau (INRA Nancy) for this work done specifically for the Centre for Strategic Analysis.

forests of 980 mm, against 859 mm for non-forested regions, a surplus of 14%;

- preferential use of good quality water sources, even if remote from distribution regions, by water treatment plants, from a relatively high contribution of forest masses. For example, 46% of the water consumed in Paris comes from catchments around Sens, Provins and Fontainebleau, which are regions quite rich in forests[1];
- the existence of quality standards for raw water to be treated. Exceeding these standards may oblige significant investment in water treatment to find alternative resources;
- a strong economic valuation of natural untreated water sources (bottled water is sold at about 100 times the price of tap water). An example can be cited in the case of the Société des eaux de Vittel *(Vittel Water Company)* which draws 1.3 billion bottles a year from a 5,000 hectare watershed, or about 260 m³ per hectare. This company invested 24 million euros in 7 years to prepare 3,500 hectares on which it carries out agricultural activities[2]; or about 1,000 Euros per hectare per year, an investment which, compared to the annual production of water, amounts to 1.52 Euros per m³.
- this estimate does not take into account uses of high quality water other than the supply of potable water which, as we have already seen, only represents a small part of the 3,200 m³ per hectare produced each year. Qualitatively, many aspects illustrating the importance of physical and chemical water quality could be cited: the effect of forest cover to moderate the temperature of water courses also conditions its use for cooling nuclear power plants (see the 2003 crisis) as well as the structure of fish stocks (and hence the value of recreational fishing); the role of riparian forests in limiting contributions of fine sediments, pesticides or fertilizer components to rivers is also accepted (see the recent report from CSPNB, 2007) and the negative consequences of eutrophication on economic activities such as tourism or oyster farming are known. We do not have quantitative data in this field to establish a reference value but we will point out a simple calculation showing that a valuation of this water at a few cents per m³ is sufficient to justify doubling the value that we propose.

[1] Source: www.eaudeparis.fr.
[2] See the site www.gesteau.eaufrance.fr/spip/spip.php?article46.

*The functions of protection*

In mountainous regions, but also in regions such as Normandy where the habitat is concentrated in valleys, the protective role of permanent vegetation cover against floods, avalanches and mudslides is recognized. The great floods of the nineteenth century in the South-east, resulting from massive deforestation and which led in 1860 to the first law on mountain reforestation, are still remembered. In the Mediterranean region, the protective role of a diversified and well maintained forest against fires is also known. We refer particularly to Lavabre and Andreassian (2000) for qualitative developments on these questions.

**Estimates of the value of this service (table VII-11) are few and highly variable however, and the methods used are not always explained and justified.** Thus, one might question the use of expenditures intended to protect the Mediterranean forests (Ref. 10) as a measure of the value that society places on these forests: we might as well consider it to be a negative externality, that is, a cost induced by the existence of these forests. Similarly, it is not said that expenditures for restoration of mountain land (ref. IFEN, 2005) are calculated from a model of economic efficiency. These estimates range from a few eurocents for Canadian forests to 1,400 euros per hectare per year for Swiss forests.

Methodologically, the study by Biao *at al.* (in press) on the forests surrounding Beijing may be cited, which propose approaching the protection function from the maximum volume of water stored by the forest floors, so helping to limit the effects of floods (while on impermeable ground this volume would run off immediately). It proposes evaluating the volume from the cost of annual amortization of a storage structure (dams, ponds), implicitly assuming that this structure does not have other values (irrigation, fishing, etc.).

Applying this method, with a volume stored at saturation of about 300 m³ per hectare and amortization values[1] of the order of 0.1 to 0.4 euros per m³ (Loubier *et al.* 2005; Tardieu, 2005; Guinaudeau, 2009), we obtain a valuation of this service in a range of 30 to 120 euros per hectare per year.

[1] This figure is approximate since it varies considerably depending on the surface area of the storage basin. It depends much more on the perimeter and depth of the project (price of dykes) than on its volume. Guinaudeau (2009) proposes, for an off-thalweg reservoir in the South West of 0.4 to 1 million $m^3$, an investment value of 4 to 6 €/$m^3$ stored, leading, with an amortization at 4% over 25 years, to a value of 0.3 to 0.4 €/$m^3$/yr.

**Table VII-11: Estimated values of forest protective functions (in dollars or euros per hectare per year)**

| Study | Function | Value | Reference |
|---|---|---|---|
| Turkey | Watershed protection | $ 7.4 | 1 |
| Turkey | Not specified | $ 46 | CDB, 2001 |
| France, Mediterranean | Wildfire and erosion | € 30 | 10 |
| France, Mountain | Protection | € 8 | IFEN, 2005* |
| Switzerland | Ground protection | € 1,360 | 14 |
| Canada | Watershed protection | € 0.06 | 12 |

* Estimate based on the budget for mountainous land restoration: 25.4 million Euros for about 3 million hectares.

In addition to water storage, reducing river sediment load may be an important component of these protective functions. The partial filling of Lake Serre-Ponçon, related to the presence of many non-reforested eroding lands upstream and the physical pollution of the Berre Pond by clay contributions from the Provence Canal are examples of this phenomenon that ought to be better evaluated in economic terms.

To draft an economic approach to this protection against erosion, we could distinguish at least three aspects, **by considering, as for carbon storage, that it is legitimate to attribute these avoided damages to the credit of ecosystems that contribute to:**

- **the "private" and local cost of land loss.** Léonard *et al.* (2009) suggest an average value of the order of 1 mm per year, or about 13 tonnes per hectare, but values ten times higher might be observed on bare soil slopes, such as wine growing areas. There is no economic model linking this loss to a possible reduction of different services, but if we consider replacement cost, i.e., replacing this lost land, for which bulk delivery prices are from 10 to 15 Euros per tonne, this leads to an average value of this service of about 150 Euros per hectare per year. This figure is consistent with that cited by Le Bissonnais *et al.* (2002) for Alsatian viniculture, namely a cost of 114 to 380 Euros per hectare per year for regularly rebuilding land on upper slopes.
- **the "ecological" cost of this loss**, that is, the consequences of exporting terrigenous contributions to other ecosystems. We know, for example, that fine elements may plug gravel zones of river beds and prevent the reproduction of trout and salmon, rendering the river sterile in the absence of any chemical pollution or physical barrier to fish movements. The consequences, particularly in terms of recreational fishing, can be significant;
- **the socioeconomic cost**, that is, the consequences of catastrophic events such as mudslides, for individuals and communities. Le Bissonnais *et al.* (2002) identified, for the 1985-2001 period, 5,579 natural disaster orders related to mudslides, with a marked increase since 1993. These events have damaged more than 30,000 buildings and it would probably be possible to use data from insurance companies to estimate the cost of this damage, to which must be added other nuisances, such as suspending the supply of potable water or repairing roads.

**Due to the small number of studies and the localized nature of these functions** (this function can no doubt be ignored for lowland forests of temperate zones while it may take high values in mountains, and even in areas of rolling hills such as Haute-Normandie), **it seems to us to be difficult, while emphasizing its potential significance, to propose an average reference value at this time and we prefer to leave this question to more specific studies.**

Regarding the role of biodiversity, we would emphasise that the example of a diverse and permanent plant population itself ensures a diversified root system in the entire floor, increasing resistance to erosion.

*Other protection and regulation services*

Among other protective services in the MEA typology, **the question of benefits for human health** has been the subject of only a few works. Willis *et al.* (2003) cite a study on the role of forests in the absorption of micro-particles and sulphur oxides. The impact study is limited to proximity effects, that is, to the lower frequency of hospitalization and death related to atmospheric pollution in forested regions. The resulting figure is low (200,000 to 11 million pounds in 2002, or, for all British forests, less than £1 per hectare per year) but it must reflect the fact that many forest regions are sparsely populated (and therefore the potential service is not used) and that the benefits at greater distance have not been measured.

Another service that has not been specifically studied but which may represent an important item in terms of long term ecological capital is the **role of forests in soil formation.** It is known that tree roots may descend into contact with bedrock, or even penetrate fractures, and thereby contribute to soil formation. As an example, Sicilian farmers, to regain ground after a lava flow, begin by planting broom-plant, the roots of which crush slag, then pine, the needles providing a first humus and, finally, vines (De Wever and Reynaud, 2009). **In a context where soil loss is perceived as a major threat especially in intertropical regions, economic studies attributing a value to this function of pedogenesis are therefore to be encouraged.** To establish an order of magnitude, Léonard *et al.* (2009) suggest a value of pedogenesis of the order of 0.1 to 0.02 mm per year, or between 0.3 and 1.3 tonnes per hectare per year. At the current price of topsoil delivered in bulk (10 to 15 € / tonne), we obtain a range of 3 to 20 € per hectare per year.

### *d. Cultural services*

The MEA includes these services in its aesthetic, spiritual, educational and leisure aspects. Many estimates are available in this field, focusing on recreational value (use values) and based on prices either revealed (methods of displacement and other costs) or declared (willingness to pay).

There is a considerable dispersion in the estimates (Table VII-12), but this variation is primarily related to attendance rates, from the great boreal forests of sparsely populated countries (Canada, Scandinavia) to forests highly exploited by tourism or a peri-urban population. To illustrate this point, we have shown in the table for each country the average number of inhabitants per km² of forest.

Study 15 (Willis *et al.* 2003) is particularly illuminating in this area: by measuring willingness to pay for a recreational visit, the authors find values of the order of £2 per visit (from £1.66 to £2.75 depending on the methods) for visits involving a journey of more than 10 miles and £0.9 Pounds for nearby visits. Aggregation of these estimates provides an average value of £134 per hectare per year for all British forests, but the forests of England, which receive 320 visits per hectare per year, reached a recreational value of £350 per hectare per year, while the Scottish forests, with only 17 visits, only manage £19 per hectare per year.

Moreover, most studies rely solely on travel costs and in this case provide rather homogenous values per visit, generally between 1 and 3 dollars (2 euros for the IFEN French study). However, Rauch (1994) or Lebreton and Vallauri (2004) propose assimilating a visit to the forest as a cultural activity and they attribute an additional value equivalent to a seat at the cinema, valuing the visit at 4 to 6 euros. Likewise, Zandersen *et al.* (2009), in a meta-analysis of 26 studies conducted in different European countries, obtained an average value of 4.52 euros per visit, which seems significantly higher than travel costs alone.

**Table VII-12: Estimated recreational forest values (in dollars or euros per hectare per year)**

| Study | Value | Reference | Density* |
|---|---|---|---|
| Turkey | $ 0.1 | 1 | 695 |
| Canada | $ 15 | 12 | 8 |
| Scandinavia | $ 15-20 | 11 | 28 |
| United States | $ 35 | Kriege, 2001 | 95 |
| United States | $ 88 | *Id.* | *Id.* |
| Italy | $ 77-85 | CDB, 2001 | 838 |
| France, | € 120 | 10 | ? |
| Mediterranean | $ 77 | Lebreton, 2004 | 393 |
| France, total | € 126 | IFEN, 2005 | *Id.* |
| France, total | $ 250 | CDB, 2001 | 639 |
| Ireland | $ 214 | CDB, 2001 | 770 |
| Germany | € 216-777 | 14 | 591 |
| Switzerland | $ 2,290 | Turner *et al.*, 2003 | 2 157 |
| Great Britain | £ 134 | 15 | *Id.* |
| Great Britain | | | |

*In inhabitants per km² of forest.

Without choosing this option (especially since payment for visiting the forest would certainly reduce visits), we can make two corrections to the IFEN estimate:

- travel costs should be updated. The IFEN study used an average cost of 0.24 euros per kilometre but a value of 0.40 euros would today be a minimum (see the official scale from the tax administration). This results in a value of 3.5 euros per visit;

- it seems necessary to take into account the fact that two thirds of French forests are privately owned and so are often less accessible to the public[1]. We propose to use a "visitable" area of only 50% of the total.

We then obtain, basing ourselves on the IFEN study which assessed the usage rate at about 17.9 visits per resident of France per year (not including visits for gathering, hunting or fishing[2]), a valuation of the forests open to the public at about 405 € per hectare per year.

**We therefore propose to use an average reference value for France of about 200 euros (which corresponds to the total expenditure incurred with respect to all French forests, averaging 58 visits per hectare per year) and to attribute a weight coefficient related to usage rates, ranging from zero for inaccessible forests to values of 5 to 10 for peri-urban forests** (see the case of English forests, reaching 320 visits per hectare per year).

These estimates only cover travel costs. For certain leisure activities, expenses incurred may be much higher. Thus, with regard to hunting, we would estimate the economic value of this activity in France using all the expenditures made by hunters (equipment, ammunition, permits, travel, etc.). This gives (Scherrer, 2002) a total value of 1.7 billion euros for the whole of French hunting in 1992 (Table VII-13), that the author suggests updating to between 2 and 2.5 billion euros in 2002. **Based on the hunting territory (36.3 million hectares for all of France according to this study, or 26 hectares per hunter), this leads to an average estimate of 55 to 69 € per hectare per year for production of "recreational hunting" and we can consider that the forest is certainly above this average value.**

However, from this value negative externalities related especially to big game (destruction of some habitats, effects on cultures, automobile accidents) should be subtracted. Overall estimates in this field do not seem to exist and, additionally, some of these externalities are taken on by the hunters. One of the most important positions, unsupported by the hunters, is probably that related to road accidents: Bourget *et al.* (2003) estimates 150 million euros as the overall cost in 2002 for insurance companies arising from collisions between vehicles and large animals.

[1] However, a survey by SCESS in 1999 (*in* MAP, 2006) indicates that 86% of the owners forest of over 1 hectare (representing 72% of the surface area) claim to allow free access.
[2] Gathering activities were assessed in the provisioning services part. Hunting is discussed later.

**Table VII-13: Assessment of expenditures by hunters in 1992 (in millions of euros)**

| Item | Expenses |
|---|---|
| Regulatory expenditures and insurance | 182.9 |
| Hunting rights | 281.1 |
| Weapons and ammunition | 217.2 |
| Equipment | 80.9 |
| Dogs (purchase, maintenance) | 665.2 |
| Travel and miscellaneous expenses | 320.7 |
| TOTAL | 1 748 |

*Source: Sherrer (2002)*

Regarding other recreational services, we note the study by Willis *et al.* (2003), which proposes to estimate a "contemplative" value based on the willingness of the British to pay for a view of the forest from their home. This willingness to pay is estimated at £269 per household, which, converted using a fairly complex process, provides, for all of Great Britain, a value of £150 million per year, or about 40% of the recreational value related to visits. Similarly, Pearce (2001) cites different studies that estimate a 3% to 6% increase in the value of houses when they have a view of a forest. It is however difficult to state this value per hectare of forest, since the areas responsible for this "landscape" effect are difficult to specify and undoubtedly limited. The role of biodiversity in modulating the value of these cultural services has not been specifically studied but it seems fairly evident that ecological and landscape diversity influences the number of visits to a forest region;  single-species plantations of trees of the same age are certainly less attractive than mixed forests.

### *e. The case of habitats and biodiversity*

We now need to address, in a particular way, the issue of forest biodiversity. Many studies have attempted to evaluate this function directly (Table VII-14). Note that the Canadian figure (from a U.S. study) is based solely on the savings in pesticides for the control of forest pests by a diversified avifauna. These values given to biodiversity and conservation of forest habitats are most often derived from contingent evaluation approaches and should therefore be considered with caution. We have particularly seen that they often paradoxically lead to a willingness to pay that is less strong than for a single emblematic species, more evocative than the concept of biodiversity, especially due to the respondents' difficulty in perceiving these concepts.

**Table VII-14: Estimated values for forest biodiversity and habitats (in dollars, pounds sterling or euros per hectare per year)**

| Study | Value | Reference |
|---|---|---|
| Turkey | $ 2 | 1 |
| France, Mediterranean | € 23* | 10 |
| France, total | € 22.8 | IFEN, 2005 |
| Switzerland | € 22 | 14 |
| United States | $ 15 to 144 | Krieger, 2001 |
| Great Britain | £ 8.4 to 98 | Willis *et al.*, 2003 |

* Data rounded from IFEN, 2005.

Recognizing this difficulty, Willis *et al.* (2003) have used a collective approach based on focus groups by which they seemed more able to developed a reasoned opinion[1]. They get a willingness to pay of about £1 per year per household for conservation of ancient semi-natural forests, against only £0.35 for replanting more diversified forests after replanting coniferous forests. But they hesitate with another sensitive issue, already mentioned, when it comes to relating these values to surface area. The question is indeed what the public is likely to pay for such operations. Thus, depending on whether we consider that only the Scots or, conversely, all British, are attached to ancient Scottish forests, the value obtained for their biodiversity rises from £8.4 to £98 per hectare per year.

Moreover, it seems that many problems of double accounting arise at this level. In particular, timber production or $CO_2$ sequestration incorporate the effects of good pest control associated with high biodiversity. Similarly, functions of subsidiary produce or recreation will also reflect this dimension.

**More generally, we have seen that biodiversity has conditioned nearly all ecosystems and we have proposed that it be (at least in terms of usage value) evaluated through these services. We therefore need to be consistent with this option and do not propose here a specific valuation for biodiversity.**

**This option leaves completely open the question of estimating non-use values.** As we have seen, contingent evaluation methods are the only ones that help to give non-use values, particularly values for existence, but the distinction between the two types of value (use and non-use) is not generally made. This distinction is undoubtedly easy for noteworthy elements of biodiversity, located far from urban areas, such as endangered species of large mammals in tropical forests; it is possible in this case to consider that the willingness to pay of residents interviewed from metropolitan France mainly reflect values of non-use. In the case of temperate forests and, more generally, common biodiversity of our territory, use values, immediate or deferred, real or potential, undoubtedly constitute an important part of the declared willingness to pay and more detailed studies would be necessary to specifically identify the values of non-use. **In addition, the "non-use value" and non-use of a service should not be confused at any given time.** In particular, we have introduced the notion of "maximum credible value" of a service in a given location to take into account the possibility of a change over time for use of a service.

### *f. Summary*

Table VII-15 summarises the reference values we have put forward.

If these different values are simply added (we have previously discussed the limits of this practice[2]) and without taking into account the different services for which we have emphasized the potential significance but could not evaluate, we arrive at an average

---

[1] « Focus groups allow people more time to consider and discuss the various aspects of biodiversity in forests, compared with individuals' responses in a CV or CE questionnaire survey ».

[2] The main questions seem to be those of double accounting, which we have tried to avoid, and negative interactions between different services, for example between tourism and hunting, more difficult to take into account.

reference value for temperate French forests of 968 € per hectare per year, with a minimum (excluding the production function and excluding tourism) of at least 450 € per hectare per year, or at least four times the value of wood production alone.

This value is higher than most of the studies that we have cited. It results, first, in a more exhaustive consideration of services (especially carbon sequestration and storage) and, secondly, the importance of the recreational function. But it is to be noted that only recreational functions in high population density regions or protective functions in fragile regions could lead to values at or above this amount, hence the need to develop this outline with more detailed work, based on the "socio-ecosystems" typology that we have proposed and which we will now demonstrate the feasibility of.

Table VII-15: Proposed reference values for various ecosystem services of French forests (in euros per hectare per year)

| Services | Proposed value | Comments |
|---|---|---|
| **Extraction services** | | |
| - wood | € 75 (€ 75 to € 160) | According to method of appraisal (stumpage or post-extraction) |
| - other forest products (excluding game) | € 10 to € 15 | |
| **Regulation services** | | |
| - carbon sequestration | € 115 | € 360 in 2030 |
| - carbon storage | € 414 (€ 207 to € 414) | € 650 to € 1,300 in 2030 |
| - other atmospheric gases | Not rated | Lack of reliable quantitative assessments |
| **Regulation services (continued)** | | |
| - water (annual quantity) | € 0 | Assuming no major effect of forests on annual hydrological balance |
| - water (flow regulation) | Not assessed | Lack of relevant studies |
| - water (quality) | € 90 | |
| - protection (erosion, floods) | Not assessed | Lack of relevant studies |
| - biodiversity | Not assessed directly | Assessed through other services |
| - other regulation services (health, etc.) | Not assessed | Lack of relevant studies |
| **Cultural services** | | |
| - hiking (excluding hunting and subsidiary produce) | € 200 (€ 0 to € 1,000) | According to use rate |
| - hunting | € 55 - 69 | Negative externalities to deduct |
| - other cultural services | Not assessed | Lack of relevant studies |
| TOTAL* | **approx. € 970** | |
| (min.-max.)** | € 500 to over € 2,000 | |

* Taking the indicated value or the average of the indicated range.

** By simply adding minimum and maximum values.

## *g. Total updated value*

For an average hectare of forest, having the characteristics of Table VII-15, we therefore obtain a total update value of about 35,000 euros (2009), this being for:

- carbon sequestration with annual volume of 3.6 tCO2/$_2$year, an updated value (until 2050, since the Quinet report made no assumption beyond that date) of about 6,300 euros;
- destocking of 90 tC, or 330 $tCO_2$ euros, about 10,600 euros;
- other services, estimated on average at 440 euros/year, a total updated value of about 18,000 euros.

### *h. Spatialization possibilities*

For temperate forests, it appears straightforward to propose reference values at a spatial scale descending, *a minima*, to the level of a *département*, or even a forest stand:

- the national forest inventory provides a details cartography of stumpage volumes, for different types of forest (see MAP, 2006), which could serve as a basis for both assessing wood extraction services and the function of carbon sequestration and storage. For other forest products (fruits, flowers, mushrooms), a spatialization by frequency of occurrence in forest stands would be sufficient, possibly with specific regional corrections for valuable products such as truffles;
- regarding the water cycle, precipitation data from *Météo France* are spatialized at a $km^2$ scale and maps are available for all catchments of potable water supply. It would therefore be possible to cross-reference the data to assess the contribution of each forest stand to the provision of quality water;
- regarding protective functions, an identification of public forests exists having protection of the physical environment as its main function. It covers 6% of these forests, mainly in mountain and coastal regions. This identification should be extended to all forests, infrastructure protection and to developing economic studies on the value of this service, as we have noted;
- regarding cultural services, data regarding use rates is probably easier to spatialize since the vast majority of visits are close by. The average population density in a 50 km radius therefore gives an acceptable substitute. Similarly, data from the *département* is available for the number of hunters and for hunting tables that give good spatial data at this scale.

## 5.5. Some points on permanent grasslands

Although covering about 10 million hectares[1], or 18% of the national territory, permanent grassland[2] has been the subject of far fewer economic studies than forests. Attention may be given to the following few points, confined to non-market services.

**There are currently no specific data available on subsidiary produce** (mushrooms, flowers, aromatic herbs) **and hunting.** These are undoubtedly the most significant items and we refer the reader to estimates made for forests (from 4 to 49 € per hectare per year, depending on the method of calculation).

**The function of carbon sequestration has been the subject of several specific studies** (Soussana *et al.* 2004; Seguin *et al.* 2007) that underline the great diversity of values depending on the mode of grassland management, with a negative effect of intensity. In addition, raw carbon sequestration figures should be corrected by the

[1] 8.2 million hectares of agricultural area and 1.8 million hectares excluding UAA in 2007 (source: AGRESTE).
[2] The concepts of "permanently grassed areas" and "permanent grasslands" should be distinguished in a more detailed analysis.

emissions of other greenhouse gases which, in the case of grasslands, can be significant: nitrogen oxides where there are large inputs of nutrients, methane emissions from ruminants. This leads us to propose a net value of 0.2 to 0.4 tonnes of carbon per hectare per year for low intensity grasslands, which would give us, with the value we used of 32 €/t of $CO_2$, a reference of between 23 and 47 € per hectare per year for 2008. Note the highly asymmetric character of storage and release functions – the reversal of permanent grassland leads within the first 20 years to a flow of carbon to the atmosphere of about 1 tonne per hectare per year.

Regarding **carbon storage,** we can consider the atmospheric part as negligible. However, the stock of carbon in the soil is considered similar to that of forests, about 70 tonnes per hectare and representing long-term storage when the grassland is preserved. With a rate of return on capital of 4%, this function can be valued at about 320 € per hectare per year (160 € per hectare per year for a rate of 2%).

For the **water cycle,** we propose the same approach as for the forest, limited mainly to aspects related to the physical and chemical water quality. For grasslands that are fairly extensively managed, it seems we can use the same values as for forests, i.e., a reference of 90 € per hectare per year. However, as with the forest, further studies would be needed to evaluate the protective functions (combating erosion, flood control). We know that, especially in the mountains, grasslands play an important role in holding the snowpack.

**Data on biodiversity relates above all to the role that flowering plant diversity plays in maintaining insect population beneficial to cultivations,** especially pollinators. The INRA report (see Le Roux *et al.* 2008) particularly stresses the beneficial effect of the presence of grasslands on nearby crops. This is an externality that should be taken into account beyond its own effect on grassland productivity. The most often cited figure is that of Robinson *et al.* (1989), which assess this beneficial effect of pollinators at 15 billion USD for the entire US territory. On the basis of UAA correspondences between the United States and France (approximately 10), this would give about 1.5 billon USD for the entire national territory, or 30 USD per hectare on average, which would approximately be doubled to obtain 2008 values.

A more recent study conducted as part of the European ALARM program (Gallai *et al.* 2009) proposes a global estimate of about 150 billion euros per year (2005 value) or about 9.5% of the total value of plant production. The same calculation applies in the case of France, leading to 2 billion euros in 2005 (author's statement). The 2007 figures (plant production value: 30.7 billion euros) give an average figure of around 40 € per hectare per year. Taking into account a contribution by permanent grasslands much higher than that of urbanised areas, forests or annual cultivation, figures for 2008 of around € 60 to € 80 per hectare per year could be put forward.

Table VII-16: Draft of reference values for the various ecosystem services of French permanent grasslands (in euros per hectare per year)

| Services | Value proposed | Comments |
|---|---|---|
| Extraction services | | |
| - livestock products | Not assessed (market) | |
| - subsidiary products (excluding game) | € | |
| Regulation services | | |
| - carbon sequestration | € 23 to € 47 | |
| - carbon storage | € 320 (€ 160 to € 320) | |
| - other atmospheric gases | Not assessed | |
| - water (annual quantity) | € 0 | |
| - water (flow regulation) | Not assessed | Lack of relevant studies |
| - water (quality) | € 90 | |
| - protection (erosion, floods) | Not assessed | Lack of relevant studies |
| - pollination | € 60 to € 80 | |
| - biodiversity | Not assessed directly | Assessed through other services |
| - other regulation services | Not assessed | Lack of relevant studies |
| Cultural services | | |
| - hiking (excluding hunting and subsidiary produce) | Not assessed | Lack of studies, less than for forests |
| - hunting | € 4 - 69 | Negative externalities to deduct |
| - other cultural services | € 60 | Landscape amenities |
| TOTAL* | approx. € 600 | This is only one order of magnitude |

Note that compared to these values, honey production (about 90 million Euros per year[1]) seems very minor. Regarding other subsidiary insects or other species (birds, rodent-predatory mammals, etc.) likely to contribute to an integrated crop protection, we have not found an economic approach to this function, which may undoubtedly be in opposition to maintaining populations having a negative impact.

Finally, with regards to **cultural services,** it is clear that grasslands contribute to the forming of popular landscapes, particularly in highlands or bocages, but we have not found studies in this field. Similarly, the calculations we made for hunting (about 60 € per hectare per year) could also be applied to them.

**In concluding this brief overview, it appears possible to state, at the present time, by limiting to carbon sequestration and storage services, quality water production and maintaining the populations of subsidiary insects, a total value of non-market services of around 600 € per hectare per year for grasslands with good biodiversity of which, as for forests, more than half would be related to carbon storage and sequestration.**

[1] About 25,000 tonnes at 3.6 €/kg in 2005 (GEM-ONIFLHOR survey 2005).

# 6. Elements for a cost-effectiveness approach

On the basis of previous studies, it would appear possible to assess ecosystem services of temperate regions at levels ranging from 1,000 to 2,000 € per hectare per year. Even if, as we noted in Chapter I, the main objective of these reference values is to provide data for the socio-economic calculation, we wanted to see if these values could serve as the basis of a cost-effectiveness approach, considering two questions in particular:

- **if used for management policies, are such values, based on an estimate of benefits, likely to be an influence for reconsidering changes in the use of territory unfavourable to their biodiversity and ecosystem services?** We distinguish the case of changes in land use in the broad sense (forest-grassland or forest-cultivation or grassland-cultivation transition) and the case of transport infrastructure and urbanization;
- **otherwise, would the sums involved allow the financing of restoration activities in other places, to effectively compensate for damages?** This leads us to consider the costs of restoration planned by various countries.

## 6.1. Are the proposed reference values an incentive?

We provide here only a few ideas, for further discussion.

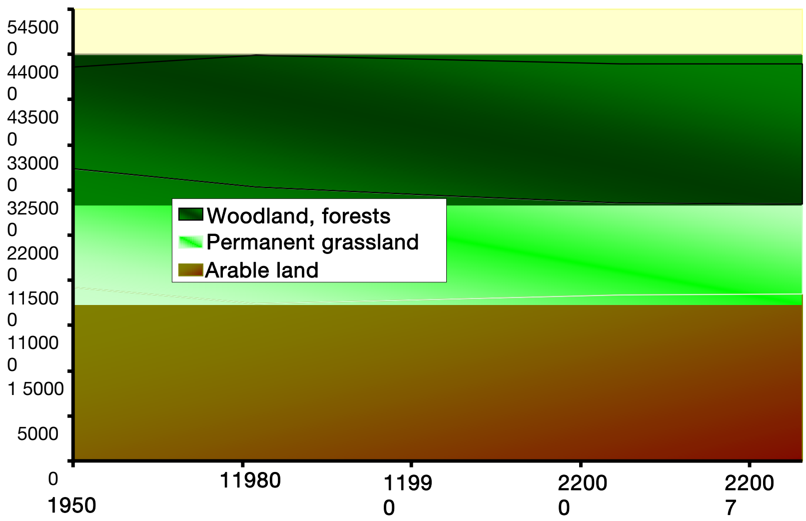
### *a. Conversion to more anthropised agricultural land*

Regarding the change of land use to produce other living resources, the main question that arises in temperate zones is the annual cultivation of areas previously under permanent plant cover (grasslands) or wetlands (drainage). This development is a strong trend since 1950 (figure VII-2) but if considered at the national level it has been accompanied by the development of woodlands, mitigating these negative effects in terms of ecological services, remaining at the global scale.

However, if until the 1960s woodland and farmland still under grass was increased to reach in total 26 million hectares in 1970, or 47% of the national territory, after this date only the forest continued to increase, which did not offset the considerable erosion of areas still grassed, with the two receding by close to 10% since this date (Dussol *et al.* 2003; MAP 2006).

This global scale does masks regional disparities, with situations that may be local issues. Thus, the Poitou-Charentes region lost 134,000 hectares of its grassed area between 1989 and 2007, or about 35% of the total, while arable land increased by 87,000 hectares and woodland remained unchanged.

Figure VII-2: Changes in land use in France since 1950
(Vertical axis: thousands of hectares. Horizontal axis is non-linear)

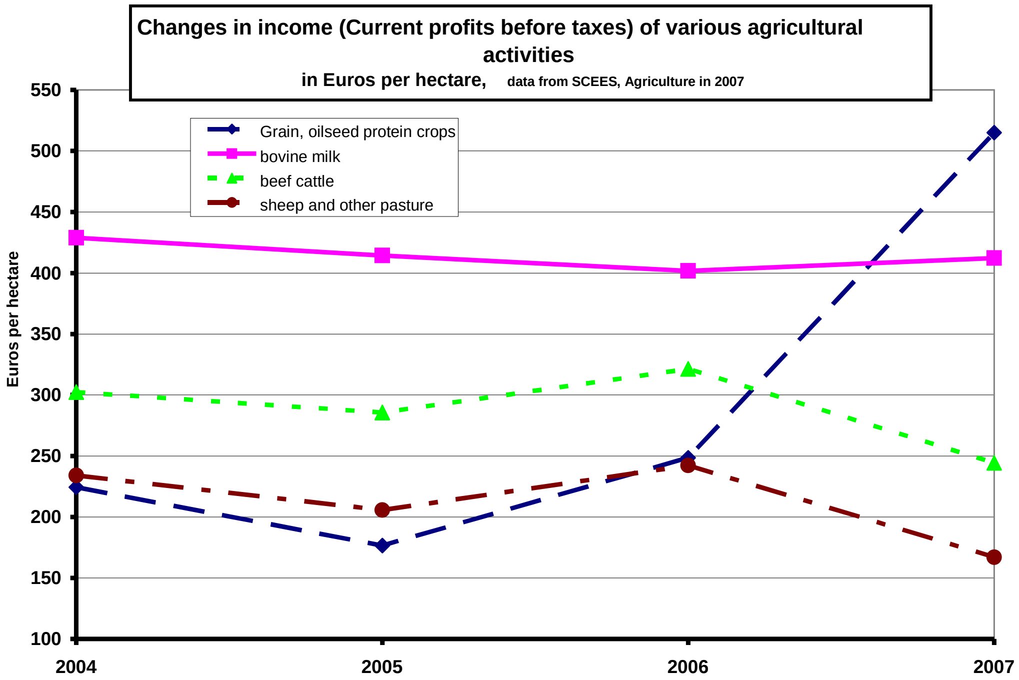



*Source: B. Chevassus-au-Louis*

The situation could be much more worrying in the future. In fact (Figure VII-3), recent changes in agricultural prices have dramatically changed the economic viability of major crops and ruminant livestock areas. The outlook for primary and even second generation energy crops may also be attractive and influence, as is already the case, reduction of grasslands for the benefit of annual crops. **However, if we examine the benefit differential between these two types of activity, we find that it does not exceed 200 € per hectare per year, 2007 being without a doubt exceptional. However, this value is significantly lower than our estimates for non-market services obtained for forests and grasslands. There should, of course, be a similar study carried out for highly cultivated areas where not all the ecosystem services vanish**[1]**, but it should be said that a compensation for services at this level of 200 Euros would effectively check this tendency of land use change.**

[1] We know, for example, that there is lively debate regarding the net contribution of energy crops on the reduction of greenhouse gas emissions, since the article of Nobel laureate Paul Crutzen proposing a reassessment by a factor of 5 of the nitrous oxide emissions of these crops (Crutzen *et al.* 2007).

Figure VII-3: Changes in annual revenue per hectare for different types of production (current profit before taxes, in current Euros per hectare)




*Source: INSEE, 2008*

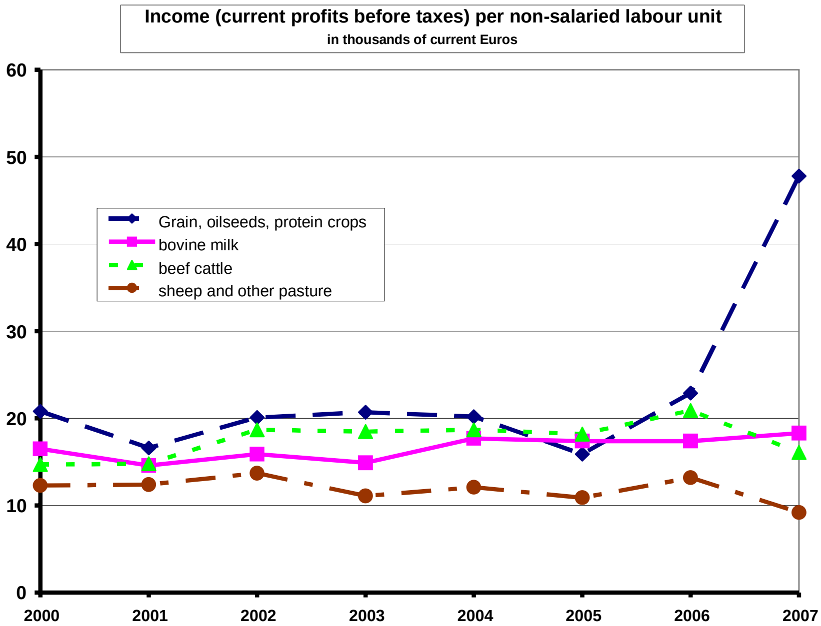
Such compensation would also be balanced with penalties incurred by France for non-compliance with European directives on the environment (Water Framework Directive, Nitrate Directive).

However, the analysis should be refined, noting that the criterion for land use change is more geared to income per person than income per hectare. It appears in fact that if income per hectare was not favourable for field crops until 2007, the income per non-salaried labour unit has for a long time, with the exception of 2005, been higher for these activities, and this difference has sharply increased since 2006 (Figure VII-4). But this observation does not invalidate the idea that a differential compensation in favour of permanently grassed areas[1] in the range of 200 to 300 € per hectare per year would be likely to counteract these changes in use.

Is such compensation far-fetched? Arguably this principle is already accepted, particularly in Europe with agri-environmental measures.

[1] In this summary study, we have included permanently grassed areas with ruminant livestock activities. We must bear in mind that some of these cattle breeders, including dairy farms on the plain, are partly based on field crops (especially corn silage).

Figure VII-4: Changes in annual revenue per non-salaried labour unit for different types of production (current profit before taxes, subsidies included, in thousands of current euros per full time labour unit)




*Source: INSEE, 2008*

Thus, we have seen that in Switzerland (Günter *et al.* 2002) support in the form of agri-environmental measures in 1999 were around 877 € per hectare per year (of a total assistance of about 2900 € per hectare per year, including farm price support). Within the European Union, funds used for this purpose were more modest: agri-environmental measures (the "second pillar") cover about 21% of the territory and are on average 80 € per hectare per year, with an additional co-financing by the country of around 50%. These figures vary greatly from one country to another, both for the affected areas (from 99% of the UAA in Finland to 3% for Greece) and for the funds used per hectare (from 29 euros for France to 246 euros for Italy). France, with 24% of its territory benefiting from these measures, is slightly above the European average, but it is the lowest in terms of assistance per hectare (Oréade-Brèche, 2005).

But we must also consider the amounts paid under the first pillar of the CAP, including SPEs (Single Payment Entitlements), which are calculated on the utilisation area and represent several hundred euros per hectare (Table VII-17). These payments are subject to conditions on environmental impact control (eco-conditionality) but, as the Table shows, are currently not discriminating between the different types of agricultural activities. It would therefore be enough[1] to increase the "environmental content" of this support (which is not a problem under WTO rules) to have a significant effect on land use.

[1] We are of course aware of the political difficulties of such a shift but recent developments in grain prices may make it easier.

Table VII-17: CAP assistance (in euros) per hectare of UAA (²) in 2006

| Type | 1st pillar | 2nd pillar | Total |
|---|---|---|---|
| Field crops | 357 | 10 | **367** |
| Bovine milk | 292 | 62 | **355** |
| Beef cattle | 325 | 91 | **416** |
| Bovine milk and beef | 337 | 43 | **380** |
| Other herbivores | 185 | 124 | **310** |
| Mixed livestock | 347 | 23 | **370** |
| **Total** | 327 | 42 | **369** |

**We show, however, that the conditionality of aid was implemented to promote "best practices" for a given agricultural activity, and not to induce changes in activity.** Still within this context, we have even seen that, in the case of grasslands, ecological services may vary greatly according to livestock practices and it would be possible to take greater account of these aspects in the SPEs.

For information, Table VII-18 also shows the amount of key measures to support forests (see details in the Appendix). The total amount is not evenly distributed between the different types of forest (private, state-owned, communal) but would be an average of 25 € per hectare per year. This figure cannot be compared with Table VII-17, since it also includes tax incentives (such assistances also exist for agriculture but do not appear in this table, which only reflects CAP subsidies).

Table VII-18: Amounts of various public subsidies for forests

| Nature | Estimated amount * {millions of 2008 euros) |
|---|---|
| Budgetary Support (Program 149) | |
| - Economic development of the sector | 38.5 |
| - Implementation of the forest regime | 168.2 |
| - Improvement of forest management and organisation | 42.9 |
| - Forest risk prevention and protection | 46.3 |
| Fiscal Expenditures | |
| - on government taxes | 71 |
| - on local taxes | 8 |
| Total | 374.9 |

* Commitment authorisations for budgetary support, estimates for fiscal expenditures

*Source: MAP*

### *b. Conversion to urban, transport and other non-agricultural uses*

**Looking now at much more drastic change of use (urbanization, transport infrastructures, etc.) the resulting increases in value may be of another order of magnitude compared to the estimates we have given for services lost, even assuming they vanish completely.** We have not conducted detailed studies to specify the value of gains for different artificial environments for the territory but it may be recalled that the transition from farmland to buildable land may multiply the land price by 100. Supporting this point of view, we also have attached a presentation of the calculation of public benefits attributed to different transport infrastructures.

In this case, the questioning is reformulated as follows:

- **should an economic tool continue to be used**, setting a reference value at least equal to the potential value gain, in order to deter such changes of use? But in this case, are we still in the economic field – with a cost-effectiveness approach – or should the regulatory tool rather be used, prohibiting use change in zones where protection is desired, and by taking from other zones a sum based on the loss of ecosystem services?
- **will sums drawn from the loss of ecosystem services allow restoration operations in other places** to ensure an overall maintenance of the "ecological capital"?
- **otherwise, should transactions based on measures other than money be envisaged?**

## 6.2. What can be said about restoration costs?

Various studies are available in this field:

- in the United States, where this restoration market has grown significantly, especially for wetlands, the OECD (2004) indicates that for the 1993-2000 period the cost per hectare restored varied from 18,000 to 247,000 dollars, these variations being mainly due to the nature of the ecosystem restored (it is more costly to restore a hectare of mangrove than a hectare of wet prairie);
- in the Netherlands, the cost of compensation defined by the "National Spatial Strategy" of 2005 varied from 10,000 to 250,000 € per hectare (Hernandez, 2006);
- the environmental assessment method of the State of Hesse (Seják *et al.* 2002) assigns to each biotope a rating of 3 to 100 points, based on a combination of 8 factors each rated from 1 to 6. The first four assess the quality of the biotope and the last four its scarcity and fragility. The overall rating results from multiplying the quality rating by the scarcity-fragility rating. Each point is assigned a set cost of restoration of 3200 € per hectare. Thus, a biotope rated at 10 points (such as arable land) has a restoration cost of 32,000 € per hectare, while one rated at 60 points (such as rain forest) has a restoration cost of 192,000 € per hectare;
- in the case of the revegetation of mining operations, Sarrailh (2002) cites a figure of 84,000 € per hectare for the New Caledonia nickel mines;
- in the case of the forests of metropolitan France, the ONF has developed scales for the "Silviculture Works Roadmaps" which integrate the costs of planting, maintenance and possible protection from game until an average forest tree height allowing spontaneous growth is reached. The 2007 values (Michel Badré, personal communication) were, for beech, from 2,600 € per hectare (regeneration from natural seed) to 4,200 € per hectare (planting) and, for oak, from 3,700 € per hectare (regeneration) to 6,000 € per hectare

(planting). An average value of 5,000 to 6,000 € per hectare could be adopted for establishing by planting a mixed deciduous stand.

It is interesting to compare these costs with the amounts that may be required as a single payment based on these reference values and related to a development permanently negating virtually all the ecosystem services of a given area (for example, establishing a paved parking space in a forest zone). If we take the reference value of approximately 32,000 € per hectare that we obtained for temperate forests (excluding timber value), we find that this figure is of the order of magnitude of the restoration costs that we have just indicated, although in the lower part of the range.

Further studies are necessary to clarify these points, but it appears at first glance that restoration costs can sometimes be substantially higher than the amounts calculated for loss of ecosystem services caused by degradation of the environment. **Hence the need to examine other approaches, based on other means of accounting than money, to establish a comprehensive conservation strategy without net loss.**

Before discussing this point we should examine in detail the procedural issues related to any exchanges.

## 7. Procedural issues

Although the working group, in its own composition, had drafted initial reference values for coral reefs, temperate forests and grasslands, we should revisit several points about the estimation and, especially, the use of these values, with particular reference to the issues discussed in Chapter III on the legal foundation.

**The working group would like to make it absolutely and unequivocally clear that the use of its work should not be a pretext to allow the degradation of biodiversity to establish itself, to put in its place a "destruction permit", with the idea that it would be possible to compensate for this degradation by a monetary transaction.**

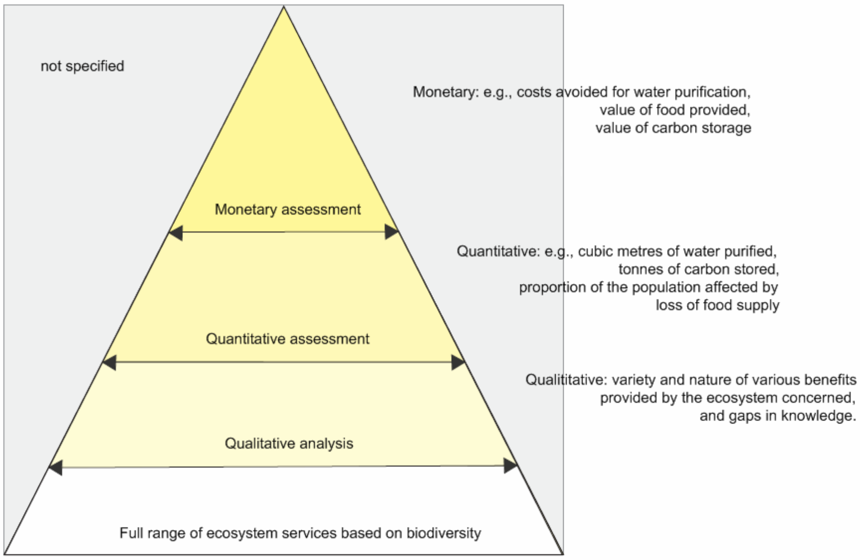
**It also wishes to remind the reader that the approach to biodiversity by use values for ecosystem services linked to ordinary biodiversity, while it allows monetary estimates to be proposed for socioeconomic calculations for public choices, strongly limits the issue:** firstly, we have repeatedly seen the contingent nature of these use values, associated in particular with the greater or lesser extent of their use at a given time and place; secondly, we have seen that certain services, potentially important, were often not assessed and, finally, that the values of non-use have remained largely neglected. Figure VII-5, drawn from the TEEB report (2008), summarizes the effect of this limitation quite well.

**In practise, the principle of an approach in three successive stages, OBVIATE, otherwise MITIGATE and finally COMPENSATE, must continue to prevail,** namely that the developer must show that it has made every effort, economically acceptable, to obviate impacts on biodiversity, otherwise to mitigate them as much as possible, before such transactions can be considered.

**Therefore, the working group considers that the examination of issues to be discussed and the effective implementation of adequate procedures to deal with them is a prerequisite for any practical use of the estimates proposed.**

We are going to discuss each of the issues related to setting reference values one by one, and then those associated with their use.

Figure VII-5: Diagrammatic representation of the field of monetization
for all ecosystem services




Source: P. ten Brink, seminar «The Economics of the Global Loss of Biological Diversity», March 5-6, 2008, Brussels

*Source: TEEB (2008)*

## 7.1. Estimation: the need for a recognized procedure

**In order that they can play their role, reference values must be recognized by the various stakeholders in territorial management as common references to clarify any potential conflicts between alternative opinions.** Given the complexity of technical aspects for setting these values, this recognition will not rest – or at least not only – on a critical review by all stakeholders (including "ordinary" citizens) of this technical content. It must therefore be based on procedural legitimacy, that is, on the fact that the process used to set these values is considered satisfactory by society, just as the authority of the thing judged resides in scrupulous adherence to a democratically adopted code of procedure.

Indeed, as we indicated earlier in this Chapter, **setting overall reference values for all services of an ecosystem needs to go one stage further compared to the "basic" values proposed by the experts for these services. It is at this stage that the question of taking into account uncertainties in estimating the value of different services must in particular be determined,** and, when the consequences of underestimating a service appears to be more severe than overestimating it, it is not obvious that using the average of available estimates is the best way of handling the information. Questions raised previously regarding the accumulation of values and the relevance of a simple summation should also be examined. Finally, the question of taking into account the long term and contingent nature of use values at a given time should be examined and included in political and ethical considerations at least as much as strictly economic ones.

**This approach cannot rely solely on experts and must be based on a known and recognized process.**

This implies in particular that:

- the composition of the group defining these reference values and the method of designating its members are defined, or at least approved by a recognized body (Office parlementaire d'évaluation des choix scientifiques et technologique ? Conseil économique, social et environnemental ?);
- the process for setting these values is precisely described and observed, and takes place openly;
- procedures for debate and clauses for periodic revision of these values are provided for.

**Another important point that we have mentioned several times involves the territorial level appropriate for setting such values.** Two questions should be distinguished in this area:

- **the question of the optimum decision level; that is, the level of highest political responsibility capable of effectively managing biodiversity and ecosystem services.** If we can exclude community level, as evidently too small, and at the other extreme the national level (due to the extreme diversity of socio-ecosystems and the weak political legitimacy in this field), the question of how appropriate are the European, national or regional levels deserves consideration. Moreover, the concept of regional level may be understood in the sense of Administrative Regions but also involves the definition of "ecoregions", that is, large biogeographic zones, like the scope of activity of River Basin Authorities or National Botanical Conservatories. However, **the Working Group considered that it was not within its mandate to pronounce on this question, also found in many other areas of public action;**
- **the question of the spatialization of values**, that is, setting values variable in space, depending on the ecological and socioecological specifics of the territories. This question is different from the previous one, in that a European national authority may well engage in spatialization at a much finer scale In this case, **the Working Group places great emphasis on the need for such spatialization.**

We will look again at this question by examining the conditions of use of such values.

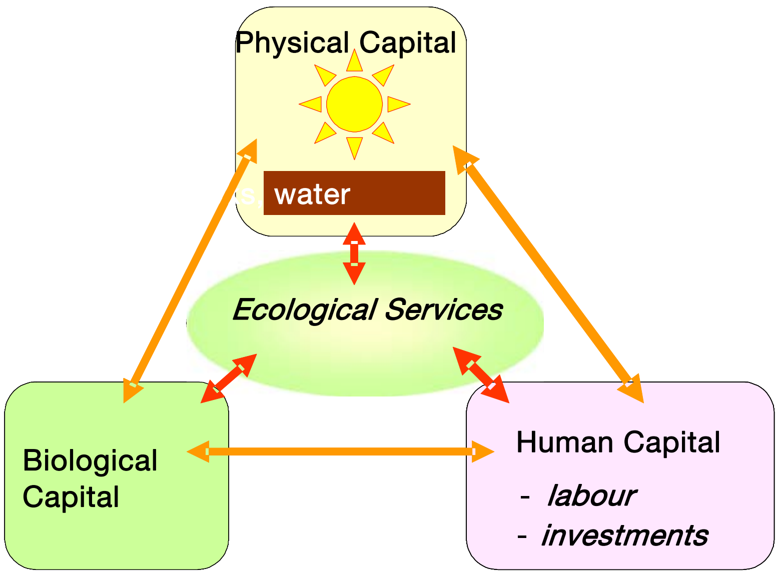
## 7.2. Use of reference values

As mentioned previously, the purpose of reference values is not to serve as the basis for compensation of private actors. However, the relatively high values that we obtained in the preceding chapter for the ecosystem services of ordinary biodiversity (forests, grasslands) raise questions about the legitimacy of public compensation of private actors managing these areas, compensation which these actors will undoubtedly be claiming.

This question can be approached in two ways, one theoretical and the other pragmatic.

**On a theoretical level,** we can start with the simple representation in Figure VII-6 for the various "capitals" which are invested in the production of ecosystem services, namely, the "geophysical" capital (solar energy and mineral, water, rocks and atmosphere resources), "biological" capital (biodiversity) and "human" capital (financial capital, skills, labour). These different capitals are, furthermore, interactive: thus, the biological capital will contribute to modulating the supply of nutrients by the physical capital and this more or less abundant supply will influence the human capital invested (supply of fertilizer).

**Figure VII-6: Schematic representation of contributions of various capitals to ecosystem service production**

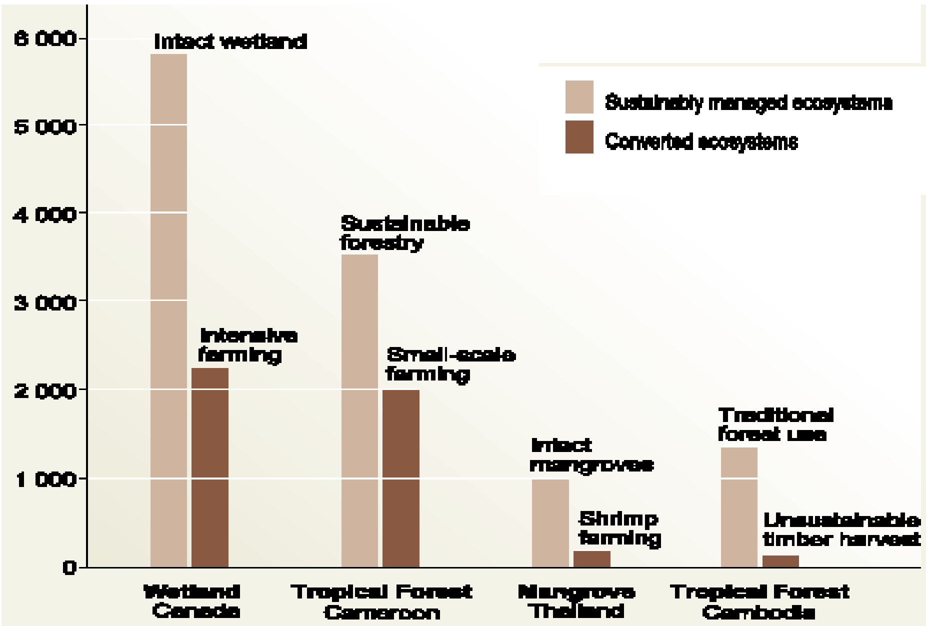



*Source: B. Chevassus-au-Louis*

According to this diagram, **the various capitals should be compensated according to their contribution to the production of ecosystem services.** In particular, in different concrete situations we need to examine how and to what extent the human capital invested leads, through "good practices", to an increase of ecosystem services and to only compensate strictly for this marginal contribution.

**Such models for estimating the specific human capital contribution should be developed but already, on this simple basis, the claim for compensation for human capital in line with the value of all of the ecosystem services can be set aside.** It may be noted that, unlike the other two capitals, human investment is neither necessary nor necessarily beneficial to the production of ecosystem services - compensation of human capital (beyond compensation for extraction services by the market) could sometimes by negative! This is the "development paradox" of ecosystems for which the *Millennium Ecosystem Assessment* has provided some examples (Figure VII-7); in many cases ecosystems that are modified to develop extraction services see their total production of ecological services reduced.

Figure VII-7: Total production of ecosystem services (in USD /ha/yr)
in sustainably managed ecosystems or
ecosystems converted to more intensive production of marketable goods




**On a practical level,** we point out firstly the unrealistic nature of compensation for all ecosystem services: although disputed, the work of Costanza *et al.* (1997) provides an order of magnitude showing that the entire world's wealth would undoubtedly be insufficient. Services that are key and susceptible to be threatened in a given location are those that should be focused on. This brings us then to the previous discussion on the cost-effectiveness approach. It is indeed then necessary to set the fraction of ecosystem services that could be compensated on the basis of changes that are to be encouraged or discouraged. This can therefore lead to either not (or only partially) compensating for these services, when the ecosystem concerned does not seem to be threatened by unfavourable development (e.g. the case of French forests, generally increasing in terms of area, although there should be vigilance regarding the qualitative trend in terms of their services), or conversely, in certain situations, to commit greater amounts than the marginal contribution of human capital to production of these services. This distortion is only possible because it is unlikely that the physical and biological capital will demand their due!

This question is discussed in detail in the recent FAO (2007) report on compensation to farmers for environmental services, which also examines questions of the effectiveness of these compensations, depending on the recipients selected and payment mechanisms used. **The report also, in another form, takes up the debate about the patentability of life:** what are the respective contributions of natural capital (through the evolution of species), the successive generations of farmers (through domestication), breeders (who created the modern varieties) and, finally, the biotechnology firms (offering genes conferring original properties) in the performance of a genetically modified variety? Is it acceptable that, through a patent, these biotechnology firms appropriate the essence of the "added value" provided by these different contributors that we have identified? In this notion of "acceptable", there is, in addition to ethical considerations, the distinction between a theoretical approach (calculation of the specific contribution of the different actors) and a pragmatic approach (what compensation mechanism is the most effective for the production of varieties adapted to farmers' needs ?).

## 8. Non-monetary approaches: compensation

As we have mentioned, **monetization is only expected to be helpful if there is a very broad market for exchange and we envisage a substitution of items for well-being that may not seem commensurable.** This is why the demand for a monetary evaluation is aimed primarily at socioeconomic studies in the preparation of public investments which have to take into account these diverse elements (time savings, health effects, improvement of economic attractiveness of the territory, environmental impacts, etc.). However, if the exchanges are possible in a more limited zone and include entities having a certain similarity e.g. two wetlands in the same region, it would seem to be feasible to define other metrics allowing for a possible "barter".

Another argument in favour of this approach is that it is probably easier, with the current state of the science, to obtain ecological expertise to directly estimate if two ecosystems –one at risk of degradation and the other resulting from restoration work – could be considered as similar, in terms of biodiversity and ecosystem services, than through monetary estimates which we have seen are complex.

### 8.1. The American approach

This compensation approach originates, generally speaking, with the 1972 *Clear Water Act* (Géniaux, 2002) in the United States. Initially, any owner wishing to establish a development likely to have impacts on a wetland must, in order to be authorised to develop, either obviate these impacts and prove this has been done, or prove that it would be impossible to obviate them, but minimize them. It is apparent that these requirements for simple minimisation have in fact led to continued degradation of wetlands. Hence the "*No net loss*" policy, defined in 1987 by the *National Wetland Policy Forum,* according to which, if impacts are not sufficiently minimized, the owner must carry out an "on-site compensation" (restoration or creation of a nearby wetland area at least equivalent) or, where impossible, an "off-site compensation". In the latter case, it uses a banking system of coordinated exchanges, "*mitigation banking*".

We refer to Géniaux (2002) and Trommetter (2008) for a detailed presentation of this system that can be summarised as follows. A compensation bank is a private or mixed operator which will acquire and restore or improve or even create wetlands at its own expense. On this basis, a public regulating body allocates it a certain number of "credits", expressed in units of surface area, which will depend on the quality of the ecosystem restored (in terms of scarcity, biodiversity, etc.). From the demand side, the operator needing to compensate for the consequences of a development will be allocated a number of "credits" taking into account the surface area affected and the extent of changes. It will then acquire from a compensation bank a number of credits equal to this number of debits. **Compensation is therefore a number of hectares of ecosystems and not a monetary value.**

**We emphasize again that this recourse to compensation is not intended to cover all potential damage to the environment: it continues along the logic of "obviate-mitigate-compensate",** that is, that the developer must show, to be authorised to have recourse to compensation, that everything economically acceptable has been done to obviate and otherwise mitigate impacts.

Furthermore, we emphasize that "**off-site" compensation, if it is in an ecologically equivalent zone, is not necessarily a last resort compared to on-site compensation** - rather than seeing a mosaic of smaller restored ecosystems, operations of "ecological consolidation" may be envisaged, focusing on coherent aggregates e.g. ecological corridors.

It is public bodies (the CORPS – US Army Corps of Engineers and the EPA – Environmental Protection Agency) which validate the quality of compensation and may impose the restoration of areas larger than the areas impacted, to take into account the uncertainties of restoration effectiveness. Thus, the OECD report (2004) indicates that from 1993 to 2000, 165 $km^2$ of restoration was carried out to compensate for impacts on 95 $km^2$ of wetlands.

Similar approaches were then developed in Australia, Canada, Brazil and Switzerland. In the European Union, this system is anticipated for Natura 2000 sites and was also developed in the Netherlands as part of the "National Spatial Strategy" (see D4E, 2005 and 2007a).

## 8.2. The case of France

In France, the obligation to compensate for the environmental impact of road infrastructures or housing was written into the law of 1976 but never actually implemented. Several pilot experiments related to industrial pollution were conducted (D4E, 2007b). It is the conversion into Law 2008-757 of the European Directive 2004/35/CE on environmental responsibility that gives new impetus to this approach, described in detail in III.2.1.

This Law states that "*Measures for remediation of damages affecting the waters and the species and habitats mentioned in 2° and 3° of I of Article L. 161-1 aim to restore these natural resources and their ecological services to their original state and to eliminate all risk of serious harm to human health.* ***The original state*** *designates the state of natural resources and ecological services at the time of damage, which would have existed if the environmental damage had not occurred, estimated using the best information available.* ***Primary remediation*** *designates any measure by which the natural resources and their services listed in the first paragraph return to their original state or approach thereto. The possibility of remediation by natural regeneration must be considered. When primary remediation fails in this return to the original state or to a state approaching thereto, supplementary remedial measures shall be implemented in order to provide a level of natural resources or services comparable to that which would have been provided had the site been restored to its original state. They may be implemented on another site, the selection of which shall take into account the interests of the populations affected by the damage.* ***Compensatory remediation*** *measures shall compensate for intermediate losses of natural resources or services occurring between the damage and the date at which the primary or supplementary* *remediation has produced its effect.* ***They may be implemented on another site and may not result in financial compensation.***

In this context, the *Caisse des Dépôts et Consignations* launched the "*CDC Biodiversité*" compensation fund in February 2008 with 15 million euros that will be managed by the *Société forestière* and which has begun to search for land in Alsace and Aquitaine (see III.1 for a more detailed presentation of this initiative).

### 8.3. Monetization and compensation

To summarize, **the compensation system presents three major differences compared to a monetary assessment of impacts that would give rise to an exonerating levy:**

- **it is clearly situated in an after-the-fact perspective, that is, after the efforts to obviate and mitigate impacts have been performed, while the monetization that we have presented is essentially situated in a before-the-fact perspective to identify the best choice;**
- **ecosystem damage must be compensated for by improvement of other ecosystems and may not be exchanged for other elements of well-being.** This is therefore in the sense of "strong sustainability" (non-substitutability of environmental capital);
- **compensation is effected on the principle of equivalence in kind** (service-service, resource-resource, without net loss) **and can therefore lead to exchange imbalances (in one direction or another) in monetary terms.**

However, three valid criticisms of monetization remain in comparison with this approach:

- **the difficulty in establishing equivalence on the basis of sound science.** Implementing a medium-term monitoring procedure with developers' responsibility (of the "decennial guarantee" type) to verify the ecological functionality of the development is one possible response to this criticism. A similar measure was proposed as part of the "Grenelle II" Bill;
- **the ever present risk of seeing this possibility of compensation favouring the fact of continued degradation of biodiversity** "in good conscience";
- **not taking into account the indirect effects of development.** For example, although the direct impact of a new transport infrastructure on biodiversity may have been compensated at the construction site, this infrastructure may favour urbanization of new areas, becoming more accessible, or increasing numbers of tourists, with negative consequences for biodiversity in these zones.

Let us point out in conclusion that the **procedural questions that we have raised about the reference values also apply to the practice of compensation, as we will now see.**

## 8.4. A market to regulate

Since the units of biodiversity are likely to be used as part of "transactions" between individuals or corporations, several issues already discussed in Chapter III arise regarding regulation of this transaction.

**Firstly, how is the decision in principle on substitutability to be made; that is, the decision whether to authorise this transaction?** We discussed this issue in particular in the case of the presence of remarkable elements of biodiversity the future of which has to be examined according to specific procedures, other than those based on ecosystem services. **Examining, by hearing all sides and in an appropriate forum, whether the remarkable elements of biodiversity have been dealt with satisfactorily, is therefore a necessary precondition for the implementation of a "transaction".** More generally, we have also discussed the need to explain the "sustainable development model" underlying this transaction: do we accept a substitution with any elements of well-being whatsoever, not related to the environment, for example replacing a polluted beach with a pool (theory of weak sustainability) or do we require there to be a substitution by improvement of other ecosystem services in other parts of the territory, in order to preserve the overall ecological capital?

The second issue that we discussed previously, for reference values, is that of **refining the values of biodiversity units, in particular when expressed in services per unit area**, by more accurate considerations on the area of influence, the location of anticipated developments and the nature of services affected.

**This also raises the question of the designation of beneficiaries (or agents) of this transaction,** that is, those who can make an offer of substitution for services lost: should they be left to develop an open and competitive market, in which the highest or best offer will be selected? Or should other criteria be used instead and this market be reserved for private operators? In the extreme, should the public power, guarantor of the citizen's well-being, be given a monopoly?

**Finally, we come back to the question of the geographic zone in which this transaction will be authorised,** that is, the zone within which the gains and losses of ecosystem services will be pooled. If we take as an example the Water Agencies, all financial resources collected in the catchment area for which they are responsible should be used to improve the overall quality and quantity of these water resources (which does not mean that the Agencies distribute this resource evenly throughout their territory). Without detailing the arguments, we consider a similar option, that of the definition of "areas of solidarity" that are relatively limited for ecosystem services, to be worth defending for political, sociological and ecological reasons. This does not mean that reference values cannot be defined on a larger scale.

With this need for regulation accepted, **the question is raised of the authority responsible for this regulation.** To respect the principle of separation of powers, we might suggest distinguishing between the "legislative" (those who set the reference values), the "executive" (those who implement the transactions) and the "judiciary" (those who regulate)[1].

## Conclusions

1. Although the issue of biodiversity and ecosystem services is significantly more complex than climate change, it is possible, for a cost/benefit approach, to provide economic estimates of ecosystem services related to ordinary biodiversity.

2. It would seem feasible to use these estimates to urge reconsideration of certain "moderate" changes of territory use, such as the development of annual crops to the detriment of surfaces with permanent plant cover, and they are of a similar order of magnitude to the costs of ecosystem restoration that are considered to be equivalent in ecological terms.

3. However, four sensitive points in this approach have been identified:

- the difficulty of approaching, even approximately, the value of remarkable biodiversity, which leads us to recommend using the economic approach in this case in only a very subsidiary manner;
- the contingent character of these estimates, since they focus on use values, likely by definition to vary depending on the intensity of these uses. We have seen for example that in the case of wood production, which is only partially exploited in France, in the production of potable water, which only uses 3% to 4% of forest water production but could increase in the future, or in tourism, which is far from saturating the forests' hosting potential. This leads to taking account of this long-teerm potential in reference values estimates;
- the lack of empirical studies on certain services that may represent *a priori* significant economic value, such as protective services or those related to the modulation of watercourse flow rates;
- these benefit oriented approaches do not appear strongly dissuasive on their own compared to high profit infrastructure development (urbanisation, industrialisation, transport) and in this case it seems to be difficult to move to a regulatory approach, at least in the areas that we wish to protect.

4. Moving from "base" reference values, based on economic analysis, in line with different methods and different ecosystem services, to global reference values for a given ecosystem, adapted and used in an appropriate manner in different practical situations, appears to be a complex but unavoidable process, for which we have tried to identify the major steps:

- first of all, the "contextualisation" of reference values in a practical situation should combine valid procedural qualities for the entire territory (rigour, transparency, revision clause, debate) with a willingness to take into account the specific local contexts, both ecological and socio-economic;

[1] Note, however, to inform the debate, that such a principle is not really implemented in the case of River Basin Authorities, where Basin Committees are involved in these three steps and then perform an "integrated" management of the process.

- a second significant conclusion, an independent "regulator", capable of acting on major choices (is substitution allowable? If so, between which actors? Have remarkable elements of biodiversity been taken into account satisfactorily?) would appear to be indispensible if we do not wish to give way to an untamed "commoditization" of nature, justified by economic analysis.

5. Finally, we have shown that, especially if the geographical scale of management were to be limited as we propose, approaches based on equivalence in kind could be more pertinent – and possibly the source of less conflict – than those based on monetization, particularly for the practice of compensation.

**In any case, it should be noted that a transaction, whether based on money or equivalence in kind, could be the final step of the process "obviate – mitigate – compensate".**

## Bibliographic References

Alfter P. (1998), « Recherche sur les biens et services non-bois de la forêt suisse. Quantification et essai de valorisation dans le cadre d'un projet de l'OFEFP »("Research on Other-than-Wood Goods and Services in the Swiss Forest. Quantification and Attempt to Valuing in the Framework of an OFEFP Project") ; *Schweiz. Z. Forstwes*, 149, 87-104, cité par Lebreton et Vallauri, 2004.

Allen B. P. et Loomis J. B. (2006), « Deriving values for the ecological support function of wildlife: an indirect valuation approach », *Ecological Economics*, 56, 49-57.

Anielski M. et Wilson S. (2005), *Counting Canada's natural capital: assesing the real value of Canada's boreal ecosystems*, Canadian Boreal Initiative and Pembina Institute, Ottawa, Canada, 79 p.

Bann C. et Clemens M. (1999), *Turkey: Forest Sector Review - Global Overlays Program Final Report*, Ed World Bank, Washington.

Beaumais O., Laroutis D. et Chakir R. (2008), « Conservation *versus* conversion des zones humides: une analyse comparative appliquée à l'estuaire de la Seine », *Revue d'économie régionale et urbaine*, n° 4, p. 569-590.

Biao Z., Wenhua L., Gaodi X. et Yu X. (2008), « Water conservation of forest ecosystem in Beijing and its value », *Ecological Economics*, in press.

Bolker B. M., Pacala S. W., Bazzaz F. A. *et al.* (1995), « Species diversity and ecosystem response to carbon dioxide fertilization: conclusions from a temperate forest model », *Global Change Biology*, 1, 373-381.

Boman M. et Bostedt G. (1995), *Valuing the Wolf in Sweden*, Report N° 110, Swedish University of Agricultural Sciences, Dpt of Forest Economics, Umeä.

Bourget J., Bracque P., de Nonancourt P. et Sapor C. (2003), *Évaluation des risques liés à l'augmentation des densités des sangliers sauvages en France*, ( Assessment of the Risks Induced by the Increase in Wild Boars Densities in France) MEDD/MAAPAR Rapport n° C 2003 T 067, 63 p.

Brahic E. et Terreaux J.-P. (à paraître), *Évaluation économique de la valeur de la biodiversité en forêt. Guide méthodologique*, (Economic Assessment of the Biodiversity Value in Forest Environment. A Methodologic Guide) Éd. QUAE, Paris.

Brander L. M., Raymond J. G, Florax M. et Vermaat J. E. (2006), « The Empirics of wetland valuation: A comprehensive summary and meta-analysis of the literature », *Environmental and Resources Economics*, 33, 223–250.

Brouwer R., Langford I. H., Bateman I. J. et Turner R.K. (1999), « A meta-analysis of wetland contingent valuation studies », *Regional Environmental Change*, 1 (1), November, 47-57.

CAS (Centre d'analyse stratégique) (2008), *La valeur tutélaire du carbone (Tutelary Value of Carbon)*, Rapport de la commission présidée par Alain Quinet, coll. Rapports et Documents, n° 16, La Documentation française, juin, 109 p.

CBD (Convention on Biological Diversity) (2007), « An exploration of tools and methodologies for valuation odf biodiversity and biodiversity resources and functions », *CBD Technical Series*, n° 28, 71 p.

Corisco-Perez C. (2006), *Potabilisation: les coûts réels de production de l'eau potable*,(*Drinking Water Treament: the Real Costs of Production*) Synthèse technique, Éd. ENGREF, Montpellier, 12 p.

Costanza R., d'Arge R., de Groot R. *et al.* (1997), « The value of the world's ecosystem services and natural capital », *Nature*, 387, 253-260.

Crutzen P., Mosier A., Smith K. A. et Winiwarter W. (2007), « $N_20$ release from agro-biofuel production negates global warming reduction by replacing fossil fuels », *Atm. Chem. Phys. Discuss.*, 7, 11191-12205.

CSPNB (Conseil scientifique du patrimoine naturel et de la biodiversité) (2007), *L'arbre, la rivière et l'homme*, (Trees, Rivers and Human Beings) Éd. MEEDAT, Paris, 63 p.

D4E (2007a), « Mécanismes de compensation: une opportunité pour les secteurs économiques, financiers et les gestionnaires de la diversité biologique »,(Compensation Mechanisms:an Opportunity for Economic, Financial Fields and for Biological Diversity Managers) *Lettre Évaluation*, avril.

D4E (2007b), « Évaluation économique des dommages environnementaux sur accidents industriels », ("Economic Assessment of Damages Inflicted to the Environment by Industrial Disasters") *Lettre Évaluation*, novembre.

D4E (2007c), *Analyse coût/avantages de la restauration d'une rivière: le cas du Gardon aval*, Collection « Études et synthèses ».( Analysis of the Cost/Benefits Induced when Restoring a River: Case of Downstream Gardon)

D4E (2006), « La valorisation des aménités du Loir »(Valuing the Loir River Amenities), *Synthèse du document de travail 06-E-01*, mars.

D4E (2005), « Mécanisme de compensation pour la conservation de la diversité biologique: état des lieux et analyse pour sa viabilité en France », (Compensation Mechanism for Preserving Biological Diversity) *Lettre Évaluation*, juillet.

De Wever P. et Reynaud J. Y. (2009), «Terroirs, les racines du vin » (Soils, the Roots of Wine), *Découverte*, n° 360, p. 13-25.

Dupouey J. L., Pignard G., Badeau V. *et al.* (2002), « Stocks et flux de carbone dans les forêts françaises » (Carbon Supplies and Flows in French Forests), *Forêt Walonne*, n° 57, p. 6-19.

Dussol A. M., Hilal M. et Kroll J. C. (2003), « 30 ans de PAC: plus de grandes cultures, moins de fourrages, autant de disparités géographiques »("After 30 Years of Common Agriculture Policy, More Large-Scale Farming, Less Fodder, Still as Many Geographic Disparities"), *Agreste Cahiers*, n° 3, p. 27-33.

European Communities (2008), *The Economics of Ecosystems & Biodiversity: An Interim Report*, 64 p.

FAO (2007), *La situation mondiale de l'alimentation et de l'agriculture. Payer les agriculteurs pour les services environnementaux (Worldwide Situation of Food and Agriculture. About Paying Farmers for the Servicesthey Provide to Environment)*, Éd FAO, Rome, 234 p.

FPF (Forêt privée française) (2008), *Les chiffres clés de la forêt privée.(Key Factors concerning Private Forests) Édition 2008-2009*, Éd FPF, Paris, 24 p.

Gallai N., Salles J.-M*.,* Settele J. et Vaissière B. (2009), « Economic valuation of the vulnerability of world agriculture confronted with pollinator decline », *Ecological Economics*, 68, 810-821.

Géniaux G. (2002), « Le *Mitigation banking*: un mécanisme decentralisé au service des politiques de *no net loss* »,( A Decentralised Mechanism to Serve "No Net Loss" Policies) *Série Actes et communications de l'INRA*, n° 17.

Guinaudeau M. (2009), « Retenues de substitution: sortir de la controverse », *Agriculture, Alimentation et Espaces Ruraux*, (Substituted Deductions: Ending the Controversy) n° 30, p. 5-6.

Günter M., Schläpfer F. *et al.* (2002), *Direct payments for biodiversity provided by swiss farmers: an economic interpretation of direct democratic decision*, Working Group on Economic Aspects of Biodiversity, Case study, Switzerland, OECD Publications, Paris, 41 p.

Guo Z., Xiao X. et Li D. (2000), « An assessment of ecosystem services: waterflow regulation and hydroelectric power production », *Ecological Applications*, 10, 925-936.

Hageman R. (1985), « Valuing Marine Mammal Populations: Benefits Valuations in Multi-species Ecosystems », *in Administrative Report*, Southwest Fisheries.

Hernandez S. (2006), « Compte-rendu du séminaire sur les mécanismes de compensation » (Report of the Seminar on Compensation Mechanisms), *Note de la D4E*.

IFEN (2005), « Les multiples valeurs de la forêt française »(The Multiple Values of French Forests), *Les données de l'environnement*, n° 105, août, 4 p.

INSEE (2008), *L'agriculture en 2007*, Rapports présentés à la Commission des comptes de l'agriculture de la Nation le 1er juillet 2008, (Reports Submitted on July 1st, 2008 to the Commission in Charge of the Nation Agriculture Budget) Éd. INSEE, Paris, 211 p.

Jactel H. et Brockerhoff E. G. (2007), « Tree diversity reduces herbivory by forest insects », *Ecology Letters*, 10, 835-848.

Johnansson P. O. (1989), « Valuing public goods in a Risky World: an Experiment », *in* Folmer H. et Ierland E. (eds), *Evaluation Methods and Policy Making in Environmental Economics*, Amsterdam, North Holland.

Keppler F., Hamilton J. T. G. *et al.* (2006), « Methane emissions from terrestrial plants under aerobic conditions », *Nature*, 439, 187-191.

Krieger D. J. (2001), *Economic Value of Forest Ecosystem Services: A Review*, The Wilderness Society, Washington D.C., 30 p.

Lavabre J. et Andreassian V. (2000), « Les eaux et les forêts. La forêt: un outil de gestion des eaux ? », (National Forestry. Forests:a Water Management Tool ?) *La lettre de Medias*, n° 13, p. 38-44.

Le Bissonnais Y., Thorette J., Bardet C. et Daroussin J. (2002), *L'érosion hydrique des sols en France*, (Water-Induced Soil Erosion in France) rapport IFEN-INRA, 106 p.

Lebreton P. et Vallauri D. (2004), *Si la forêt m'était comptée…(A Forest Tale ….)*, Rapport scientifique, Éd. WWF-FRAPNA, 23 p.

Léonard J., Le Bissonnais Y., Andrieux P. et Darboux F. (2009), « L'érosion, un acteur majeur de la dégradation des sols et de l'environnement », (Erosion, a Major Factor to Soil and Environment Decay) *Dossier INRA « Le sol »,* Éd. QUAE, Paris, p. 128-131.

Le Roux X., Barbault R., Baudry J., Burel F., Doussan I, Garnier E., Herzog F., Lavorel S., Lifran R., Roger-Estrade J., Sarthou J. P. et Trometter M. (éds) (2008), « Agriculture et biodiversité. Valoriser les synergies. Expertise scientifique collective » (Agriculture and Biodiversity. Valuing Synergies.Collective Scientific Expertise), *synthèse du rapport INRA* (France), 113 p.

Loomis J., Kent P., Strange L. *et al.* (2000), « Measuring the total economic value of restoring ecosystem services in an impaired river basin: results from a contingent valuation survey », *Ecological Economics*, 33, 103-117.

Loomis J. B. et White D. S. (1996), « Economic benefits of rare and endangered species: summary and meta-analysis », *Ecological Economics*, 18, 197-206.

Loubier S., Christin F., Aubry N. *et al.* (2005), « Comment prendre en compte la demande en eau pour l'irrigation dans un contexte de rareté et d'incertitudes sur la ressource: un exemple de combinaison d'outils sur le bassin de la Charente en France »,(How to take into Account the Demand for Irrigation Water with a Background of Resource Scarcity and Uncertain Availability: an Example of Tools Combining for the Charente River Basin in France) *Communication International Conference on Environmental Science and Technology*, Cyprus, 12-13 mai.

Magnani F., Mencuccini M., Borghetti M. *et al.* (2007), « The human footprint in the carbon cycle of temperate and boreal forests », *Nature*, 447, 848-852.

MAP (ministère de l'Agriculture et de la Pêche) (2006), *Les indicateurs de gestion durable des forêts françaises,(The Indicators for a Sustainable Management of French Forests) édition 2005*, Ed. MAP, Paris, 148 p.

MEA (Millennium Ecosystem Assessment) (2005), *Ecosystems and Human Well-Being. Synthesis*, Ed. Island Press, Washington DC, 155 p.

MEEDAT (ministère de l'Écologie, de l'Énergie, du Developpement durable et de l'Aménagement du territoire) (2008), « La valeur économique de la forêt méditerranéenne en France », (*The Economic Value of the Mediterranean Forest in France) Lettre Évaluation*, n° 17, juin.

MEEDAT (ministère de l'Écologie, de l'Énergie, du Developpement durable et de l'Aménagement du territoire) (2008), « La préservation des écosystèmes coralliens: principaux aspects scientifiques, institutionnels et socioéconomiques », (Preservation of Coral Ecosystems : Main Scientific, Institutional and Socio-Economic Aspects) *Document B2-08-051*, 28 p., et *Lettre Évaluation*, n° 18, juillet.

Montagné C. et Niedzwiedz A. (2007), « Évaluation économique des bénéfices marchands et non marchands de la forêt française »,(Economic Assessment of the Mercantile and Non-Mercantile Profits of the French Forests)*Forêt-entreprise*, n° 176, p. 30-34.

Mullan K. et Kontoleon A. (2008), *Benefits and Costs of Protecting Forest Bodiversity: Case Study Evidence*, Final Report submitted to the IUCN and EEA, 82 p.

OECD (2004), *Handbook of Market Creation for Biodiversity: Issues in Implementation*, OECD Publications, Paris, 177 p.

OECD (2002), *Manuel d'évaluation de la biodiversité. Guide à l'intention des décideurs*, (Biodiversity Assessment Manual. A Guide for Decision Makers) OECD Publications, Paris, 177 p.

OECD (2001), V*aluation of Biodiversity Benefits. Selected Stud*ies, OECD Publications, Paris, 190 p.

ONF (Office national des forêts) (2006), « Forêt et Carbone »(Forest and Carbon), *Dossier n° 05*, juin.

Oréade-Brèche (2005), *Évaluation des mesures agroenvironnementales. (Assessment of the Agro-Environmental Measures) Rapport final*, 233 p., disponible sur www.oreade-breche.fr.

Pearce D. (2007), « Do we really care about Biodiversity? », *Environmental & Resource Economics*, 37, 313-333.

Pearce D. W. (2001), « The economic value of forest ecosystems », *Ecosystem Health*, 7 (4), 284-296.

Pearce D. W. et Pearce C. G. T. (2001), « The value of forest ecosystems », *CDB Technical Series*, 4, 59 p.

Pointereau P. (2007), *L'approvisionnement en eau potable de la ville de Münich. Une collectivité engagée pour un développement durable, (Drinking Water Supply to the City of Munich. A Collectivity dedicated to Sustainable Development)* www.solagro.org/site/im_user/153_notestechmunich.pdf.

Rasool I. (1993), *Le Système Terre*, (The Earth System) Paris, Flammarion.

Rauch T. (1994), *Kapital Wald*, cité par Lebreton et Vallauri, 2004.

Robinson W. S., Nowogrodzki R. et Morse R. A. (1989), « The value of honey bees as pollinators of U.S. crops. II », *American Bee Journal*, 129, 477-487.

Sarrailh J. M. (2002), « La revégétalisation des exploitations minières: l'exemple de la Nouvelle-Calédonie »(Recreating Plant Coverage over Mining Zones: the Example of New Caledonia), *Bois et Forêts des Tropiques*, n° 272, p. 21-31.

Scherrer V. (2002), *Réinventer la chasse pour le XXIe siècle*, (Reinventing Hunting Practices for the XXIst Century) rapport au Conseil économique et social, 266 p.

Seguin B., Arrouays D. *et al.* (2007), « Moderating the impact of agriculture on climate », *Agricultural and Forest Meteorology*, 142, 278-287.

Seják J., Sovova L. et Kupka M. (2002), *Applied Evaluation of Biodiversity,* Working Group on Economic Aspects of Biodiversity, Case study: Czech Republic, OECD Publications, Paris, 29 p.

Soussana J. F., Loiseau P. *et al.* (2004), « Carbon cycling and sequestration opportunities in temperate grasslands », *Soil Use and Management*, 20, 219-230.

Stenger A., Harou P. et Navrud S. (2009), « Valuing environmental goods and services derived from forests », *Journal of Forest Economics*, 15, 1-14.

Tardieu H. (2005), « Water management for irrigation and environment in a water-stressed basin in SouthWest France », *in* M. Svendsen (ed.), *Irrigation and River Basin Management*, CAB International.

Trommetter M. (coordonnateur) (2008), *Biodiversité, agriculture et politiques publiques. Expertise scientifique collective (Biodiversity, Agriculture and State Policies),* chap 4, INRA, Paris, 108 p.

Turner K. R., Paavola J., Cooper P. *et al.* (2003), « Valuing nature: lessons learned and future research directions », *Ecological Economics*, 46, 493-510.

Willis K. G., Garrod G., Scarpa R. *et al.* (2003), *The Social and Environmental Benefits of Forests in Great Britain*, Report to Forestry Commission, Edinburgh, 36 p., www.forestry.gov.uk/website/PDF.nsf/pdf/sebreport0703.pdf/$FILE/sebreport0703.pdf.

Woodward R. T. et Wui Y. S. (2001), « The economic value of wetland services: A meta-analysis », *Ecological Economics*, 37, 257-270.

Zandersen M. et Tol R. S. J. (2009), « A meta-analysis of forest recreation values in Europe », *Journal of Forest Economics*, 15, 109-130.

# Chapter VIII

# General Conclusions

As we come to the end of this report, we will emphasise some of its major conclusions regarding the four major issues of the mission.

## 1. The socio-economic issues for biological diversity in France

These issues revolve around the – relatively recent – perception of two realities: the major and sometimes irreplaceable role of biodiversity in human well-being, including the providision of the most essential needs, and the threats it is now facing and which are causing its accelerated erosion.

With regard to the first aspect, we have emphasised, as have others, the major importance of conserving biodiversity in the present, but also, and perhaps above all, for the future. This "ecological capital", that also includes the products of the past activity of living beings[1], appears more and more to be one of the sources of tomorrow's innovations and of sustainable development. Without making any value judgement on ethical conceptions that give equal value to all living beings, **an anthropocentric and utilitarian approach, as long as it includes future generations, is in itself sufficient to acknowledge the extent of what is at stake.** Let us remember that the impression of increasing independence from nature that had marked the second half of the 20th century was essentially linked to the fact of the massive use of the products of past biodiversity – above all fossil fuels – the limits of which are now clearly seen.

With regard to threats to biodiversity, it is certainly legitimate today to consider the fight against climate change and the conservation of biodiversity as two related challenges of equal importance, inasmuch as the effects of climate change will be more or less drastic depending on the effect it has or doesn't have on biodiversity and, conversely, biodiversity is liable to modulate the extent of these changes. **But it is important to stress that the already substantial current erosion of biodiversity is due to factors that have been going on for many years:** modification and fragmentation of habitats, pollution, introduction of invasive species, desertification, overuse, etc. **It is therefore essential to get these various factors as under control as quickly as possible, because in the future climate changes could interact with them and increase the pressure they now put on biodiversity further still.**

France is involved in these questions on at least three levels: its status as a developed country, a country with internal and external environmental impacts that are

---

[1] The term "geodiversity" is sometimes used to refer to all of the natural mineral resources, many of which result from past biodiversity activity. We didn't analyse this subject in this report but we want to draw attention to this aspect which, with the exception of energy resources, receives less media attention.

particularly high and continue to increase; its capacity for influence, alone or through the European Union, to work to ensure that these issues are taken into account on an international level (G20, United States, WHO, etc.); lastly, its bio-geographic position in particularly bio-diverse zones or on the boundaries of these zones, both in Metropolitan France and in the Overseas Departments and Territories.

## 2. Inventory of scientific knowledge

We firstly had to consider how biological sciences can now characterise the object(s) to be analysed in an economic sense. This has shown the extent to which biodiversity is a complex matter which, depending on the concerns and the space and time scales involved, must be approached with specific and diversified indicators. There is no “accounting unit” similar to the equivalent tonne of $CO_2$ for climate change and it seems unlikely that greater knowledge will lead to one being proposed. But we should stress that **this does not mean that an appraisal of the state and evolution of biodiversity, on various spatial scales, is impossible. It just requires a combination of the reports from experts and various indicators, a situation found in many other areas such as social and health policies.**

As we have seen, **the introduction of the notion of ecosystem service has the advantage of facilitating an integration** and, in particular, it involves a vision of biodiversity that focuses not just on its components, but also on their multiple interactions. Another advantage is that this service approach stresses the importance of the quantitative dimension, i.e., beyond the notion of diversity in the strict sense, the abundance of the individuals contributing to these services. **But looking at biodiversity in terms of its services is also limiting, because it favours a view based mostly on the current or foreseeable - and also quantifiable - uses of biodiversity.** Even if the concept of “total economic value” includes non-use values, providing relevant and reliable estimations of these values seems very difficult. But we chose this option to allow for its pairing with the economic analysis, while suggesting that “remarkable” biodiversity entities be handled in a specific way.

With regard to the economic analysis, we didn’t hide the fact that there is a real reticence, and principled objections, to its implementation. These objections are based on at least two considerations: the first is that **the notion of the value of biodiversity has – and should keep – a much broader sense than that which economics is likely to grasp**; the second is that the **introduction of an economic value can implicitly introduce the idea of the “merchandising” of biodiversity**, i.e. exchanging it with other goods, although the goal of stopping its erosion is now acknowledged and part of the political agenda. There is also the issue of the “status” of biodiversity, with frequent reference to the delicate notion of “public good.” For this reason we thought it necessary to make explicit the bases and hypotheses of the theory of value in economics, to properly define the relevance of this approach and to emphasise the difference between “market price” and “reference value.” Moreover, **analysis of law practices has given us an approach for taking these objections into account** – through notions of procedural and regulatory legitimacy – while allowing the economic analysis to be applied.

More practically speaking, we have inventoried the range of methods developed to attribute monetary values to non-marketable goods. Beyond the fact that their theoretical bases are unequal, in their practical application these methods provide results with considerable variations for a given service, sometimes of several orders of magnitude. We have tried to determine the origin of this by distinguishing five registers:

- Firstly, **as for marketable goods, these values can vary in space and over time**, depending on the economic context (revenues), and also social and cultural factors;
- The second source of variation, which also exists for marketable goods, is related to the **information that the "consumer" has about the goods.** This issue is particularly complicated for biodiversity which often has to be "translated" into more concrete entities – species or sets of familiar or emblematic species, ecosystems that are known and particularly rich in biodiversity – to allow the people questioned to express their opinion. We should also consider the role of the various actors (experts, political decision-makers, NGO's, medias, etc.) in the forming of these representations of biodiversity; but this issue of information is one of the reasons we decided to distinguish remarkable biodiversity, for which the entities are explicit, culturally specific, and can be grasped directly, essentially through a contingent valuation – of which we have stressed the limits –, and ordinary biodiversity for which the economic value can be approached indirectly, through the ecosystem services that depend on it, and with other more solid methods (revealed preferences, substitution costs);
- The third source of variation, **the processing and pooling of individual preference data**, in the case of declared preferences, **to relate it to a given entity** (an endangered species, an ecosystem in the process of degradation, a hectare of forest, wetlands, etc.) **appears to be a particularly sensitive point** and one that is not adequately dealt with in most studies. For this reason, in the case of remarkable biodiversity, we encourage the use of the results of economic analysis in an ancillary manner only, in a deliberative context, in the light of the debate on the legitimacy of this approach and the variability of the results that it provides[1];
- **The "rate of use" of ecological services also greatly conditions the estimations obtained, but in a very comprehensible way**. We demonstrated this for water quality –according to the proportion of water provided by the ecosystems that is used for the production of drinking water – and for tourist access to forest areas. This leads us to introduce the notion of the "maximum plausible value" of a service, a value to be combined in a way that remains to be defined with its use value at a given time;
- Lastly, **issues of pooling**, whether for an evaluation of the various services for a given space or for totalling the services of different spatial units to obtain an estimation on a larger scale, **cannot be limited to simple addition; appropriate methodologies must be developed.**

[1] We should note that the use of participatory or deliberative approaches is precisely one of the approaches recommended by certain specialists of the methods based on declared preferences, to reinforce the validity of the results.

## 3. Research needs

By limiting ourselves to the most directly operational aspects, we have identified several research orientations that should be encouraged.

**In the field of biological sciences, the development of databases, monitoring systems and composite biodiversity indicators, on different spatial scales and over the entire country, appears as the basis of any policy in this field.** Several initiatives are already underway and we can only encourage their support and hope they become permanent. The establishment of ONEMA, a public establishment one of the central missions of which is the construction of a national database for water and aquatic milieus, constitutes one example of a recent initiative in this direction. The restoring and continuation of long-term observation systems and of biological field stations (particularly marine stations) should also be encouraged. More broadly, the establishment of a national biodiversity observatory, included in French governemnt's *Grenelle de l'environnement* programme, should provide an institutional framework for these developments.

We also emphasise the **importance of being able to associate in these systems information derived from state indicators and indicators of pressure on biodiversity linked to various human activities.** These pressure indicators offer two advantages compared to state indicators: first, as for concentrations of greenhouse gases in the atmosphere to analyse the process of global warming, they can measure the specific effect of policies that directly attack factors of biodiversity erosion and can take on a "control parameter" character for these erosion phenomena; also, they constitute "advance indicators", because they can detect future favourable evolutions of the state indicators. To do this, quantitative models associating these pressure indicators to foreseeable evolutions of biodiversity should be developed.

Still in the field of biological sciences, we saw the importance that the concept of ecological equivalence could take on in the future, particularly in compensation practices. **It would be valuable to develop approaches that are as explicit and transparent as possible to establish this equivalence, including its margins of uncertainty, and to implement systems to validate these approaches.**

Lastly, the emergence of the concept of ecosystem services and its use in economic analysis leads on to the need to specify the link between these services and the various dimensions of biodiversity, particularly to define the modifications of biodiversity which, for a given ecosystem, may or may not modify the scope of these services in the short, medium and long-term. In this field, experimental ecology approaches integrating this long-term dimension, not widespread in France, should be encouraged to supplement the observation systems. These approaches should be broadly multi-disciplinary, i.e. associating biological approaches with the physical chemistry of natural environments, biogeochemistry, ecotoxicology, etc.

**In the field of economic and social sciences, the challenges seem to be more in the lack of concrete work applying the available methods rather than methodological developments.** We emphasise this point because it means that the academic dynamics and incentive systems that depend on them may not be able – or at least not on their own – to make up for this lack. We saw in particular the weakness – in comparison with those for tropical environments – of studies devoted to the services of temperate zone ecosystems and the disproportion between certain highly documented services, such as the sequestration of carbon or recreational value, and others for which estimations are rare, such as protection functions. In this latter case, coupled approaches, involving specialists of the physical environment, ecology, economics and risk management must absolutely be promoted, at the risk of being partial. The issue arises for nature risks but also for human health, in its link with biodiversity and the environment, whether for modulation of presence or the effect of pathogenic agents or polluting substances.

**Another important dimension in the mobilisation of the social sciences is the analysis of appropriate procedures for the integration and sustainable management of biodiversity.** We mentioned this for the establishment of reference values and the regulation of their use, and also for the issue of the expression of preferences in contingent valuations. This must also be considered with regard to the reactions of certain actors of society to the various incentive systems that could be implemented, based on the evaluation of ecosystem services: it would be naïve to imagine that these actors behave based on a simple model of rationality; they are obviously capable of strategic behaviour, the correct anticipation of which is a condition for the success of conservation policies. In conjunction with this issue, pedagogical research on consciousness-raising, education and information to citizens, taking into account their socio-cultural diversity, should be encouraged.

Lastly we will mention the issue of the **integration of the heterogeneity of preferences with regard to biodiversity,** such as the diversity of points of view between the various actors present in a territory, or tensions that could exist between local appreciations and those of other stakeholders from outside the territory and who have other priorities. It would be quite superficial to try to present this diversity of individual preferences through an "average preference," and work at the interface between economics, sociology and political sciences could shed light on this problematic.

We end on the issue of the law and the legal "status" of biodiversity. It appears that **only one part of biodiversity** (the genetic resources of domesticated species, protected species, remarkable spaces, etc.) **has a real status that specifies the rights and obligations of public and private operators.** On the other hand, ordinary biodiversity, such as herbaceous flora, the macro-fauna of the soil and, above all, the micro-organisms of the soil and water, is considered as the private property of those who own and use the territories. In contrast, we note the wealth of legal provisions concerning ownership of the sub-soil and the riches it may contain. If, as we have amply emphasised, this ordinary biodiversity is a major determinant of ecosystem services, we can thus ask whether legal sciences should modify its "status".

## 4. Estimation of reference values

By applying the aforementioned methods, we have demonstrated, for some ecosystems present in France, how it has been possible to estimate a certain number of their services. We have also identified the services for which the assessment currently seems insufficient, particularly that of protection services. **The values that we obtained confirmed – and even reinforced – the frequently-mentioned conclusion that the total economic value of non-marketable goods from these services is at least equal to, if not much greater than, that of marketable goods.**

But our estimations should be considered as orders of magnitude. They have to be refined, supplemented and adapted to the diversity of ecological situations and also local socio-economics. **For this reason, we proposed establishing a typology of "socio-ecosystems" and developing the reference values for each of them.** The "MEA France" project that is now in progress, which works on the characterisation, quantification and cartography of the various ecosystem services, will contribute to this.

**This need for spatialisation of the reference values may disappoint those who, on the model of a tonne of carbon, have wanted national or even international references.** It could also appear to contradict the very objective of reference values which must, by definition, transcend local contingencies to create forms of arbitration that are as objective as possible. But this contradiction is only apparent, and the spatialisation of values as a function of indicators that must be specified seems unavoidable, at least in the short-term, in the case of biodiversity and ecological services. The appropriate spatial scale remains to be specified however, with the understanding that the procedure orienting the determination of these values must be defined on a national or even international scale.

We have also emphasised that the setting of a comprehensive reference value for all of the services of a given socio-ecosystem must, based on the values available for these various services, their uncertainties and the incompleteness, make the necessary choices, but that with the current state of knowledge they cannot be totally objective, **whence the importance of defining a procedure seen as legitimate for making these choices and setting these values.**

With regard to the use of these values, we mentioned in the introduction that the important but complex issue of their concrete inclusion in the various instruments of public action (regulations, taxes, incentives, etc.) and the relative efficiency of these various options was beyond the framework of this report. However, **we demonstrated, in the first analysis, that this exercise could modify the "micro-economic" hierarchy between various agricultural uses of the soil** (annual crops, permanent grasslands, forests) and, for each use, between the various modes of production management. This conclusion is particularly important in the context of current thinking on the objectives and modalities of application of the Common Agricultural Policy.

Conversely, it would seem to be difficult, again in the first analysis, to rely on these values to encourage a complete reconsideration of major changes of use involving high added value (urbanisation, transport infrastructure). **In this case, these values could however be used in the comparison of various options for a given development and justify recommendations in favour of biodiversity:** as we see from the example of some large cities, ecological engineering now has the capacity to design such systems, even in highly anthropised environments.

We have thus far been using an *ex-ante* rationale, i.e. the use of reference values as decision-making tools. However, once a decision has been made, and after examining the possibilities to be avoided and then attenuated, there is the issue of compensation for possible residual effects. We therefore considered how this principle of compensation could apply, again stressing the need for a local scale and **the importance of adapted and legitimate procedures to regulate these practices. We demonstrated that it was possible, even desirable in this case, to use a notion of**

**equivalence in kind rather than using reference values,** for which the field of application must remain chiefly the *a priori* socio-economic analysis of public choices.

## 5. Recommendations

In addition to the abovementioned research needs, the working group identified a small number of operations that could be carried out in the short-term and that would usefully supplement this report.

**1. Based on the approach of the ecological services of ordinary biodiversity, it would seem to be possible to rapidly extend the work done by the group on several "pilot" ecosystems.** The working group thus recommends, based on the model of the synthesis done for temperate forests:

- Preparing exhaustive critical syntheses of the available data (in Metropolitan France and in comparable regions) and that which is lacking for the major types of temperate ecosystems;
- Examining the relevance for the Overseas Departments and Territories of the substantial work done on inter-tropical ecosystems;
- Doing a first spatialisation in Metropolitan France, on the scale of at least one department, of the data available on the various marketable and non-marketable services, starting with forest ecosystems for which this seems theoretically feasible.

**2. To make the development of the reference values sustainable over time, to reinforce their methodological bases, and also to ensure their legitimacy in the eyes of the actors involved, the group recommends identifying (or creating?) a standing pluralistic structure** (based on the model of the *Grenelle de l'environnement* "colleges"?), basing itself on this report and also on other works in progress in this field (TEEB, work of the European Environment Agency, MEA France), responsible for:

- **defining the reference methods** (when several methods are available) for the evaluation of the various ecosystem services;
- **setting the key parameters** to be used for these calculations (value of a tonne of carbon, cost of drinking water treatments, value of a unit recreational visit, etc.), in order to homogenise the estimations of these services;
- **enriching the concept of "maximum plausible value"** (see chapter VII) and possibly proposing initial estimations;
- **Comparatively examining the approaches used for the economic evaluation of other negative externalities** (noise, atmospheric pollution) and if necessary proposing a system for the coherence of those used for the evaluation of ecosystem services.

Based on this methodological framework, which must be updated regularly, it will be possible to assign the formulation of concrete estimations to various operators (delocalised services, engineering firms).

3. With regard to **the use of these reference values, the group stresses the need to define (or to issue a reminder of, when they exist) the places and decision-making processes** which, based on the spatialised reference values:

- will estimate, for a given development, the values to take into account, considering in particular the areas directly but also indirectly impacted by a development (these areas can be variable according to the services) and the effects linked to the localisation of these areas (fragmentation of habitats, threshold effects);
- will take into account elements of remarkable biodiversity, which must, as we mentioned, involve the use of other references beyond mere economic evaluation;
- will apply the "avoid-attenuate-compensate" sequence, including definition of the legitimate compensations based on ecological equivalences in kind.

4. In order to define a truly operational objective, the group considers it is necessary to **specify, in particular in terms of indicators and reference territorial scale, the national objective of stopping the erosion of biodiversity by 2010**, and possibly defining a new medium-term objective.

5. Lastly, the group recommends **supporting and developing initiatives to make various groups aware of the stakes of socio-economic biodiversity** (schools, students, economic actors, local authorities, etc.), stressing in particular the current and future contribution of biodiversity and ecosystem services to sustainable development.

# Appendices

# Appendix I

# Mission

The Prime Minister 16 January 2008

No. 2003

Mr. Eric Besson
Secretary of State with the Prime Minister
for Prospective Studies and Evaluation of Public Policies
33 rue Saint Dominique
75007 Pris

Mr. Minister/Dear Eric,

The debate that has taken place within the framework of the *Grenelle de l'environnement* forum has reminded us that the evolution of biodiversity is at the heart of our society's environmental concerns.

The President of France is aware of the accelerating loss of biodiversity throughout the world and the associated decline in the services provided by ecosystems. For this reason, in his speech at the time of the presentation of the conclusions of the *Grenelle de l'environnement* Forum, he made a commitment that all future public decisions would take into account their cost for biodiversity.

To this end, it is necessary to have elements for objective valuation that will allow for better appreciation of the value of biodiversity and the services rendered by ecosystems. These valuation elements could move towards the determination of reference values that could guide the Government in its decisions.

Knowledge of the economic valuation of biodiversity and the services rendered by ecosystems should be improved by undertaking specific research work on this subject. On the international level, several studies have already been carried out to try to give a monetary value to biodiversity and the services rendered by ecosystems. These include the Millennium Ecosystem Assessment and the work of the Union for Nature (UICN). Other projects are on-going, particularly the "Stern-like Biodiversity Study" following the initiative taken during the ministerial meeting of the G8 on biodiversity which was held in Potsdam in March 2007 with the five emerging countries. This latter study seeks to evaluate the cost associated with the overall loss of biodiversity and to compare the cost of inaction with the cost of conservation measures.

In this context, I would be grateful if you could establish a group of French and foreign experts that will be in charge of:

- Preparing a review of scientific knowledge on the subject of the monetization of the services rendered by ecosystems and the value of biodiversity.

- Analysing the socio-economic stakes of biological diversity in France, including the Overseas Departments and Territories.
- Proposing terms of reference for possible future research.
- Estimating the first biodiversity reference values that could be used, particularly in socio-economic studies for infrastructure projects.

The work of this commission should include experts on these issues as much as possible, including the main people involved in the *Grenelle* programme. It should take into account the work already done on this subject so that the values and methodologies selected are consistent with the approaches of our European partners.

To allow these analyses to be used for the implementation of the Government's decisions, the mission must begin as soon as possible and submit its conclusions at the end of the third quarter of 2008.

Sincerely yours,

Francois Fillon

Copy to the Director General of the Strategic Analysis Centre

# Appendix II

# Composition of the group

*Chairman:*

**Bernard Chevassus-au-Louis**, general agriculture inspector, UMR GABI, genetic aquaculture team, INRA

*Vice- Chairman:*

**Jean-Michel Salles**, CNRS, UMR 5474 LAMETA, Montpellier

*Recording Secretary:*

**Jean-Luc Pujol**, scientific advisor, Department of Research, Technologies and Sustainable Development, Strategic Analysis Centre

*Main authors:*

**Sabine Bielsa**, MEEDDAT, Sétra, Bagneux

**Bernard Chevassus-au-Louis**, general agriculture inspector, UMR GABI, genetic aquaculture team, INRA

**Gilles Martin,** professor at the University of Nice-Sophia Antipolis, instructor at the IEP of Paris (Regulation Chair), Lawyer at the Bar of Nice

**Jean-Luc Pujol**, scientific advisor, Department of Research, Technologies and Sustainable Development, Strategic Analysis Centre

**Dominique Richard**, Deputy director, European Thematic Centre on Biological Diversity

**Jean-Michel Salles**, CNRS, UMR 5474 LAMETA, Montpellier

*Members:*

**Dominique Auverlot**, Head of the Department of Research, Technologies and Sustainable Development, Strategic Analysis Centre

**Didier Babin**, researcher, CIRAD

**Michel Badré**, President of the Environmental Authority Group, General Council for the Environment and Sustainable Development (CGEDD), MEEDDAT

**Robert Barbault**, Director of the Department of Ecology and Management of Biodiversity of the Museum of Paris

**Luc Baumstark**, instructor, University of Lyons, Scientific Advisor, Strategic Analysis Centre

**Gilles Benest**, France Nature Environnement, leader of the “Economy of Biodiversity” project

**Joshua Bishop**, chief economist, IUCN (International Union for Conservation of Nature)

**Jean-Jacques Blanchon**, Biodiversity, agriculture, regions and territories project, Nicolas Hulot Foundation for nature and man

**Jean-Pierre Bompard**, delegate for Energy, the Environment and Sustainable Development, CFDT

**Denis Couvet,** professor at the Museum and École polytechnique, correspondent at the Agriculture Academy of France, CRBPO, National Museum of Natural History (MNHN)

**Patrick Falcone**, Deputy to the Assistant Director of Biomass and the Environment, Department of Agricultural, Agro-Industry and Territorial Policies, Agro-Industry and Environmental Strategy Department, Ministry of Agriculture and Fishing

**Aurore Fleuret**, MEEDDAT, CGDD, Department for the Economy, Evaluation and Integration of Sustainable Development, Sub-Department for the Economy of Natural Resources and Risks, La Défense

**Jean-Pierre Giran**, Member of Parliament, University Professor

**Pierre-Henri Gouyon**, professor at the Paris Museum of Natural History

**Alain Grandjean**, member of the Ecology Watch Committee of the Nicolas Hulot Foundation

**Jérôme Larivé**, Environmental Office manager, Sub-Department for the development of the national road network, Transport Infrastructure Department, MEEDDAT, La Défense

**Hélène Leriche**, scientific advisor to the Nicolas Hulot Foundation

**Harold Levrel**, researcher, IFREMER

**Robert Lifran**, Director of research at INRA, UMR LAMETA, Montpellier

**Michel Massoni**, general engineer, Ponts et Chaussées, coordinator of the Economy and Transport Regulation Committee, CGEDD/S2, MEEDDAT

**Joël Maurice**, honorary professor at the École Nationale des Ponts et Chaussées

**Helen Mountford**, Head of division, Climate change, natural resources and environmental outlook division, OECD Environment Directorate

**Guillaume Sainteny**, Director of Economic Studies and Environmental Evaluation at the Ministry of Ecology and Sustainable Development

**Michel Trommetter**, Director of research, UMR GAEL INRA and Pierre Mendès France University of Grenoble

**Claire Tutenuit**, general delegate, Entreprises pour l’environnement

**Jacques Weber**, Director of research, CIRAD, member of the Economic Council for Sustainable Development, Lecturer at the École des hautes études en sciences sociales and the University of Paris VI-Pierre and Marie Curie, Ecology Watch Committee of the Nicolas Hulot Foundation

# Appendix III

# The indicators

**Table AIII-1: Indicators of biodiversity proposed by the IFB and recommendations for their development (new scientific strategy of the French Institute of Biodiversity - IFB, July 2008)**

| Generic indicator | Indicators proposed | Development approaches (IFB scientific group) |
|---|---|---|
| Abundance and distribution of species | - Common birds<br>- Common butterflies<br>- Aquatic birds<br>- Large carnivores | - Supplement with indicators of other functional groups essential to the functioning of terrestrial ecosystems (plants, other groups of insects, predator fish, fresh water fish, the soil).<br>- In each case, it is important to distinguish indigenous species from alien species.<br>- Annual multi-species and multi-site monitoring.<br>- The "Biodiversity Intactness Index" (Scholes and Biggs, 2005) could allow for integration of all of the data. |
| Status of endangered and/or protected species | - UICN red lists<br>- Species involved in Natura 2000, Birds directive, Habitats directive | - Adaptation to French scale, using the methods proposed by the UICN.<br>- Combination with analysis of endemic species. |
| Surface area of biomes, ecosystems, and selected habitats | - Area occupied by various types of ground occupation<br>- Area and composition of the types of forests<br>- Area of environments that are little artificialised | - Taking into account the functionality of habitats: weighting this extension by the level of connectivity/fragmentation of the various habitats.<br>- The extension of human infrastructures could be an indicator of threats. |
| Genetic diversity | - Genetic diversity of domestic animals, cultivated plants, and fish of socio-economic importance | - This indicator must also take into account the genetic distances between varieties.<br>- Adapting this indicator to the various regions of production, as regional differences have ecological/agronomic implications that differ considerably from the intra-regional diversity.<br>- Evaluations also to be done for selected wild species, particularly species on red lists, evaluating the link with the degree of threat.<br>- Examining the possibility of genetic impoverishment of the populations on the borders of the area. |
| Protected spaces | - Area of protected spaces<br>- Area of Natura 2000 sites (Bird directive and Habitats directive), adequacy of these proposals<br>- Area of Ramsar sites | - Comparing the efficacy of protection policies according to the protection statuses, the size of these spaces (National parks, regional nature parks, Natura 2000 sites, biological reserves, etc.): e.g. using the "Abundance and distribution of selected species" indicator.<br>- Having a sufficient number of observation sites in the protected spaces. |
| Nitrogen deposits | - Exceeding of critical load | - Knowing the load, a parameter that is more transparent than the "exceeding of the critical load" index.<br>- Having other indicators of the pressures from agriculture: phosphates, pesticides, etc. |

| Generic indicator | Indicators proposed | Development approaches (IFB scientific group) |
|---|---|---|
| Number and cost of biological invasions by alien species | - Total list of alien species<br>- Cost of invasions by alien species, management plans | - The proportion of alien species in populations, their functional role and the area occupied, their numbers.<br>- Judiciously defining in advance the species judged to be "invasive".<br>- Creation of black lists based on functional, logical (invasive species elsewhere) and modelling criteria. |
| Impact of climate change on biodiversity | | - The modification of the distribution could be more relevant than phenological changes,<br>- Need to establish a relationship between these displacements and the decline in biodiversity (risks of extinction).<br>- Likely need to develop these scenarios.<br>- The marine environment indicator could be: extreme observable latitudinal extension. |
| Trophic index | Marine trophic index | - One or several equivalents should be sought in terrestrial ecosystems (defoliation index, combination of indices of bird abundance, butterflies, and the state of plant formations, etc.). |
| Connectivity and fragmentation of ecosystems | Forests, wetlands and rivers, by bio-geographic region and by country | - Considering the connectivity and fragmentation of all types of occupation of ground likely to be involved in biodiversity. |
| Quality of ecosystems | Biological quality of rivers | - Having indicators that can separate the various characteristics of this quality, particularly: physical-chemical quality (nitrate content, pesticides, heavy metals, etc.), diversity, abundance of various vertebrates and invertebrates. |
| Area of forests, agricultural, aquacultural and fishing systems under sustainable management | Involving forests, agricultural systems, aquacultural systems, fishing systems | - Considering agricultural systems, areas in organic farming, or covered by various agro-environmental measures.<br>- Taking into consideration the work of Europe on "High-Value Nature Farmland".<br>- Comprehensive indicator characterizing fishing activity: could be the percentage of over-exploited species in a given ecosystem.<br>- Strictly evaluating the criteria of sustainability, in order to precisely define what is understood by "sustainable uses".<br>- Another possibility: assessment of the efficacy of so-called "sustainable" practices, for example the indicator "Abundance and distribution of selected species", and by comparing the results according to the determined sustainability. |
| Ecological footprint | Ecological footprint | - Developing indicators other than the ecological footprint, in order to: 1) separate the direct impacts on the ecosystems from the indirect impacts through global warming, 2) assess the local sustainability of this consumption by comparison with the productivity of local ecosystems. |
| Access to and sharing of benefits | Patents for inventions based on genetic resources (plant variety certificates, etc.) | - Not just focusing on the production of immediately usable ecosystem goods (food, genetic resources, etc.), but also on the benefits associated with ecosystem services, the regulation of ecosystems (flood control, contribution to human health, etc.).<br>- A cost for awaiting protection measures could be calculated. |
| Financial transfers | | - Transfers (financing of biodiversity) to be compared with the preceding indicator. The funds allocated to research, the measurement of the efficacy of research in the field (number of publications), and of training provided (number of Ph.D.'s, number of natural scientists), could be relevant. |
| Public opinion | | - A "sensitivity and involvement of the public" indicator could be indirectly supplied through the importance given by the media to the subject of "biodiversity". |

# Appendix IV

# Examples of *ex-post* socio-economic estimation

*Ex-post* estimation of the socio-economic benefit (value for the community of a project) of some major infrastructure projects.
(Source: LOTI Appraisals; analysis and table: CGEDD – Sétra)

## 1. Development of the socio-economic appraisal of an infrastructure project

It is possible to find out about the socio-economic impact of road and rail infrastructure projects through the “LOTI appraisals” that are made public. The purpose of these appraisals is to analyse and explain the discrepancies between the forecasts of the Declaration of public utility (DUP), a document released to the public, and the real observations after the opening of the infrastructure and verification of observance of the Government’s commitments. They are governed by Law n° 82-1153 of 30 December 1982 [1] on the orientation of domestic transportation (LOTI) which provides, in its article 14, for the production of socio-economic and environmental appraisals three to five years after the opening of major transport infrastructures and the decree of application n° 84-617 of 17 July 1984 [2].

## 2. Precautions for reading

The values presented are intended to give an order of magnitude for the socio-economic values of major rail and road infrastructure projects. Their heterogeneity makes it impossible to compare their respective values. To compare the net present value of several projects, we must first make sure that the following three parameters are identical for each project: the discounting date chosen, the discount rate chosen, the evaluation period.

For the cases listed in the table, the hypothesis of calculation duration for the total discounted return (TDR) is 20 years for the LGV (high speed train lines) and infinity for motorways. This difference makes it **impossible to compare LGV and road data**.

The ratios were calculated from TDR data discounted as of various dates. This heterogeneity makes it **impossible to compare the information between ratios**.

The discount rate used in the calculations of the table is 8%. This hypothesis evolved with the instructions: those in effect in February 2009 were from May 2005 [6] and recommended a discount rate of 4%. For each project, the total discounted return is calculated for a given discounting date and varies as a function of this date: **it is**

**therefore essential to have the total discounted returns for the same discounting date so that they can be compared.**

## 3. The total discounted return - TDR

Among the socio-economic indicators inventoried in these appraisals, the TDR indicated is the difference between the overall net advantages from the project and the cost of the project. It allows for an evaluation of the usefulness of the carrying out of the project for the community.

Its mode of calculation is detailed in the framework instructions and/or instructions and circulars that emerge from them and that are in effect at the time of the Declaration of public utility. For most of the cases listed in the table, this involves either the circular of March 1986 [3], or the framework instructions of 1995 [4], or the road instruction of 1998 [5]. These documents determine the working hypotheses: nature of the advantages and costs to take into account in socio-economic appraisals, calculation duration (in the table, 20 years for LGV and infinity for motorways[1]), discount rate (8% in the calculations of the table).

Table AIV-1: TDR of some infrastructures

| Name | Length (km) | Total discounted return (1) | Discounting date |
|---|---|---|---|
| LGV | | | |
| LGV Atlantic | 285 | 391 | 1992 |
| LGV Rhône-Alpes | 106 | 316 | 1994 |
| LGV Inter-connection | 102 | 187 | 1993 |
| LGV Mediterranean | 250 | 107 | 2001 |
| LGV East (DAM) | 299 | 249 | 1997 |
| MOTORWAYS | | | |
| A39 Dijon - Dôle | 35 | 277 | 1991 |
| A39 Dôle - Bourg-en-Bresse | 109 | 252 | 1990 |
| A57 Cuers - Le Cannet-des-Maures | 34 | 209 | 1991 |
| A430 Pont Royal - Albertville | 16 | 240 | 1991 |
| A837 Saintes - Rochefort | 36.5 | 113,5 | 1991 |
| A54 St-Martin-de-Crau - Salon-de-Provence | 25 | 314 | 1996 |
| A14 Orgeval - La Défense | 16 | 1 782 | 1996 |
| A77 Dordives - Cosnes | 95.5 | 111 | 1999 |

(1) In millions of euros, 2003 value. Discounting at 8% over 20 years for LGV and infinity for motorways.

[1] For LGV: historically the SNCF used a duration of 20 years with a discounted residual value. For motorways: historically the Motorways Department used a duration of 30 years, but the residual value was implicitly taken into account through a calculation of the constant advantages from 30 years to infinity.
Since the circular of 2004, the evaluation periods are 50 years for all modes of transportation.

## 4. Extraction of information

Lastly, the LOTI appraisals allow for extraction of the following information listed for LGV and motorway projects.

Based on this information, it is possible to construct indicators of total discounted return divided by the right of way of the infrastructure. By taking the discounted returns of the projects to the same discounting date and assuming a right of way of 7 ha/km for LGV and 14 ha/km for motorways, we have the following information.

Table AIV-2: Indicator of discounted return of some infrastructures

| Name | Length (km) | Total discounted return (1) | Discounting date | Total rights of way in ha (2) | Ratio of TDR/average right of way (3) |
|---|---|---|---|---|---|
| LGV | | | | | |
| LGV Atlantique | 285 | 391 | **1992** | 1 995 | **0.20** |
| LGV Rhône-Alpes | 106 | 316 | **1994** | 742 | **0.43** |
| LGV Interconnexion | 102 | 187 | **1993** | 714 | **0.26** |
| LGV Méditerranée | 250 | 107 | **2001** | 1 750 | **0.06** |
| LGV Est (DAM) | 299 | 249 | **1997** | 2 093 | **0.12** |
| MOTORWAYS | | | | | |
| A39 Dijon - Dôle | 35 | 277 | **1991** | 490 | **0.57** |
| A39 Dôle - Bourg-en-Bresse | 109 | 252 | **1990** | 1 526 | **0.17** |
| A57 Cuers - Le Cannet-des-Maures | 34 | 209 | **1991** | 476 | **0.44** |
| A430 Pont Royal - Albertville | 16 | 240 | **1991** | 224 | **1.07** |
| A837 Saintes - Rochefort | 36.5 | 113.5 | **1991** | 511 | **0.22** |
| A54 St-Martin-de-Crau - Salon-de-Provence | 25 | 314 | **1996** | 350 | **0.90** |
| A14 Orgeval - La Défense | 16 | 1 782 | **1996** | 224 | **7.96** |
| A77 Dordives - Cosnes | 95.5 | 111 | **1999** | 1 337 | **0.08** |

(1) In millions of euros, 2003 value. Discounting at 8% over 20 years for the LGV and infinity for motorways.

(2) Based on a right of way requirement of 7 ha/km for the LGV and 14 ha/km for motorways.

(3) In millions of euros per ha, 2003 value.

# Bibliography

[1] Law n° 82-1153 of 30 December 1982 of orientation of domestic transportation modified by the basic Law n° 99-533 of 25 June 1999 for the organisation and sustainable development of the territory.

[2] Decree n° 84-617 of 17 July 1984. Issued for application of article 14 of Law n° 82-1153 of 30 December 1982 concerning major infrastructure projects, major technological choices and master plans for domestic transportation infrastructure.

[3] Letter-circular of 14 March 1986 concerning recommendations for the economic calculation and evaluation of projects in the sector of transportation and instructions concerning methods of evaluation of road investments in the open countryside and in urban areas.

[4] Framework Instruction concerning methods of economic evaluation of major transport infrastructure projects attached to the circular of 3 October 1995 of the Secretary of State for Transportation.

[5] Circular n° 98-99 and instruction of 20 October 1998 concerning methods of economic evaluation of road investments in the open countryside.

[6] Framework Instruction of 25 March 2004, updated on 27 May 2005 concerning methods of economic evaluation of major transport infrastructure projects.

# Appendix V

# Forest aid measures

We distinguish two major forms of aid: budget subsidies and tax aid.

## 1. Subsidies

These appear mostly[1] in the *Forest* programme (n° 149) of the "*Agriculture, fishing, forest and rural affairs*" project, which takes into account the multi-functionality of French forests in their economic, social and ecological dimensions. It pursues the principal goal of better management of forests in all of their functions (production, protection and social), within the framework of European and worldwide agreements for the sustainable management of forests of which France is a signatory.

The programme includes:

- development of optimal use of wood resources through improved competitiveness,
- reinforcement of the capacity of forests to resist fires and natural risks,
- promotion of forest management that develops environmental quality and the social role of our forests,
- reinforcement of the positive contribution of forest biomass in the national balance of the emission/absorption of greenhouse gases, particularly through the use of wood energy,
- support for research efforts in the forest and wood sector.

In 2008, the amount of the commitment authorizations (CA) came to 293.2 M€ and the amount of the payment appropriations (PA) was 301.6 M€.

### 1.1. Action n° 1: Economic development of the forest-wood sector

(35.8 M€ en CA and 31.3 M€ en PA)

About 12 M€ (CA and PA) were devoted to the national forest inventory to inventory forest resources throughout Metropolitan France.

Public establishments and technical centres of the forest-wood sector benefited from 20.6 M€ CA: FCBA (the organisation created from the merger of the Technical Centre

[1] Other forms of aid could be implemented within the framework of the 162 programme (Governmental territorial interventions) or through Natura 2000 contracts.

for Wood and Furniture and the French Association for Cellulose), national committee for the development of wood.

The transfers to companies involved measures of the plan to restore the competitiveness of sawmills (8.8 M€ CA), and subsidies to micro-enterprises for the mechanization of the wood harvest (4.3 M€ CA).

2.1 M€ CA were devoted to prospective studies and evaluations in the forest-wood sector.

## 1.2. Action n°2: Implementation of the forest system

(168.2 M€ CA and 177.6 M€ PA)

Most of the expenditures were for compensation payments. This payment corresponds to the expenditures undertaken by the ONF for the implementation of the forest system in the forests of local governments. This amount was stabilised in current euros within the framework of a new ONF-State contract 2007-2011 (141.5 M€).

There was also compensation for the increase in the rate of contributions for the civil pensions of civil servants (25.9 M€ in CA = PA).

8.6 M€ was used to pay for State forest work to continue the reconstitution following the storm of December 1999. Starting in 2007, these operations were self-financed by the ONF (commitments).

## 1.3. Action n°3: Improvement of forest management and organisation

(42.9 M€ CA and 51.3 M€ PC)

The operating expenses of action 3 were devoted to subsidies for public service charges paid to the National Professional Forest Cleanliness Centre (CNPPF) and to the 18 Regional Forest Cleanliness Centres (CRPF), and to the innovation, development and training action programme of the UCFF (municipalities owning forests) for a total of about 19 M€ (CA and PC).

The forest investments represented 21.4 M€ CA and 29.9 M€ PC. These operations involved the improvement of forest stands (conversion to seedling forests, pruning, thinning, etc.), the creation of forest services and provisions of the “Chablis plan” (reconstitution of stands following the storm of 1999).

Lastly, the management of regional sectors made possible the financing of actions intended to help the evolution and adaptation of forest production in the light of market demand (qualitative and quantitative knowledge of the resource, improvement of wood quality, training and information for actors): 2.5 M€ CA and 2.3 M€ PA.

### 1.4. Action n°4: Prevention of risks and protection of forests

(46.3 M€ CA, 41.3 M€ PA)

This action covered the financing of:

1. the functioning of the correspondent-observer network of the Forest Health Department (DSF);
2. operations to defend forests against fires: surveillance of sensitive forest areas, infrastructures (trails, firebreaks, water connections, etc.), public information, monitoring of scrub clearing regulations, etc.
3. the functioning of the service for the restoration of mountain terrain (RTM) which reports to the ONF and carries out public interest assignments on behalf of the State and local governments aiming to prevent natural risks and to limit harm to persons and property.
4. the upkeep of RTM forest structures.
5. the surveillance and upkeep of State dune lands.

## 2. Tax aid

### Notice

The degree of reliability of the costing of tax expenditures depends on the availability of the data needed for the reconstitution of the tax that would be due in the absence of the tax expenditures in question. Moreover, the costing of tax expenditures can integrate neither modifications of the tax behaviour of taxpayers that they induce, nor interactions between tax expenditures.

The costings presented for 2009 have been done solely on the basis of the measures voted before the submission of the draft finance law for 2009. The impact of the tax measures of this draft law on the 2009 revenues is presented in volumes I and II of the appendix "Evaluation of ways and means."

### 2.1. Main tax expenditures on State taxes (8)

(In millions of euros)

| Tax expenditures on Government Taxes contributing to the programme as the main element | | Costing for 2007 | Costing for 2008 | Costing for 2009 |
|---|---|---|---|---|
| 400108 | **Partial exemption of woods and forests and shares of interests held in a forestry unit, rural properties rented by long-term lease and GFA (farming consortium) shares**<br>Wealth taxes<br>*Objective: to help the silviculture sector*<br>*Beneficiaries: 59 200 households - Costing method: Simulation - Reliability: very good - Creation: 1981 - Last modification: 2006 – CGI (general tax code): 885 D, 885 H* | 45 | 45 | 45 |
| 520109 | **Partial exemption from real property transfer taxes for woods and forests and shares of interests held in a forestry unit, rural properties rented by long-term lease, GFA shares and the fraction of the shares of rural forestry units representative of forest properties or properties of an agricultural nature**<br>Registration fees and stamp taxes<br>*Objective: to help the silviculture sector*<br>*Beneficiaries: 1 600 households - Costing method: simulation - Reliability: good - Creation: 1963 - Last modification: 2006 - CGI: 793-1-3° and 4°, 793-2-2° and 3°, 848 bis* | 25 | 20 | 20 |

(In millions of euros)

| Tax expenditures on Government Taxes contributing to the programme as the main element | | Costing for 2007 | Costing for 2008 | Costing for 2009 |
|---|---|---|---|---|
| 110226 | **Income tax reduction for forest investments and work**<br>Income tax<br>*Objective: to help the silviculture sector*<br>*Beneficiaries: 3 540 companies and households - Costing method: simulation - Reliability: very good - Creation: 2001 - Last modification: 2006 - CGI: 199 decies H* | 4 | 4 | 4 |
| 730215 | **Rate of 5.5% for silviculture and forest operation work for the benefit of agricultural operators**<br>Value-added tax<br>*Objective: to help the silviculture sector*<br>*Beneficiaries: (number undetermined) companies - Costing method: reconstitution of tax base from data other than tax data - Reliability: order of magnitude - Creation: 2000 - Last modification: 2000 - CGI: 279-b septies* | 2 | 2 | 2 |
| 310204 | **Exceptional depreciation equal to 50% of the amount of the sums paid for the subscription of shares of forest savings schemes**<br>Corporate income tax<br>*Objective: to help the silviculture sector*<br>*Beneficiaries: (number undetermined) companies - Costing method: reconstitution of tax base from tax return data - Reliability: order of magnitude - Creation: 2001 - Last modification: 2001 - Treasury measure - CGI: 217 terdecies* | • | • | • |
| 230507 | **Taxation at a reduced flat income tax rate of 6% and a flat corporate income tax rate of 8% for capital gains made from contributions to a forestry unit**<br>Income tax and corporate income tax<br>*Objective: to help the silviculture sector*<br>*Beneficiaries: (number undetermined) companies - Costing method: reconstitution of tax base from tax return data - Reliability: order of magnitude - Creation: 1963 - Last modification: 1992 - CGI: 238 quater* | • | • | • |
| 110241 | **Reduction of taxes for dues paid to authorised union associations having as their purpose the carrying out of prevention work for the defence of forests against fires on land included in classified forests**<br>Income tax<br>*Objective: to help the silviculture sector*<br>*Beneficiaries: 2 120 households - Costing method: reconstitution of tax base from tax return data - Reliability: very good - Creation: 2006 - Last modification: 2006 - CGI: 200 decies A* | • | • | • |
| 200214 | **Increase of the declining balance depreciation for certain types of equipment of companies involved in the initial processing of wood**<br>Income tax and corporate income tax<br>*Objective: to help the silviculture sector*<br>*Beneficiaries: (number undetermined) companies - Costing method: reconstitution of tax base from data other than tax data - Reliability: order of magnitude - Creation: 2001 - Last modification: 2002 - Treasury measure - CGI: 39 AA quater* | 0 | 0 | |
| Total cost of tax expenditures[1] | | **76** | **71** | **71** |

[1] The "total cost of tax expenditures" constitutes a sum of tax expenditures for which the levels of reliability may not be identical (see "Reliability" characteristic indicated for each tax expenditure). It does not take into account programmes of less than 0.5 million euros ("•"). Moreover, in order to allow for comparison from one year to another, when a tax expenditure is non-costable ("nc"), the amount taken into account in the total corresponds to the last known costing (2008 or 2007 amount); if no amount is known, the zero value is used in the total. The significance of the total turns out to be limited because of the possible interactions between tax expenditures. It is therefore only indicated as an order of magnitude and cannot be taken as a real summation of the tax expenditures of the programme.

## 2.2. Main tax expenditures on local taxes, covered by the State (1)

(In millions of euros)

| Tax expenditures on local taxes, covered by the State, contributing to the programme as the main element | | Costing for 2007 | Costing for 2008 | Costing for 2009 |
|---|---|---|---|---|
| 060103 | **Exemption in favour of land planted with woods**<br>Tax on land<br>*Objective: to help the silviculture sector*<br>*Beneficiaries: (number undetermined) companies and households - Costing method: reconstitution of tax base from tax return data - Reliability: very good - Creation: 1941 - Last modification: 2001 - CGI: 1395* | 7 | 8 | 8 |
| **Total cost of tax expenditures** | | **7** | **8** | **8** |

# Appendix VI

# EPBRS research recommendations

Recommendations issued at the end of the meeting of the European Platform for Biodiversity Research Strategy (EPBRS), under the German Presidency of the EU (Leipzig, 5-7 May 2007), "Sustainable Use of Biodiversity".

Biodiversity and ecosystem services – Framework for the evaluation of ecosystems for the millennium (MEA) in a European perspective.

The report of the Commission of the European Communities to "Stop the loss of biodiversity between now and 2010 and beyond" (Com (2006) 216 final) stresses the importance of biodiversity in ensuring human well-being. In order to guarantee these services, the participants at the meeting considered as high priorities research on:

1. **Improving knowledge about the links between biodiversity, functions of ecosystems, ecosystem services and their dynamics, particularly in order to:**
   1.1. Better understand the contribution of biodiversity to ecosystem services;
   1.2. Better understand the influence of pressure factors such as comprehensive and political changes on conservation and the use of ecosystems;
   1.3. Identify the complex dynamics, non linear responses and abrupt or irreversible changes of ecosystems;
   1.4. Develop concepts for the accounting of the resources linked to ecosystem services, to support evaluations and management (for example, concept of Services production unit, ecosystem accounts, "bouquets" of services).

2. **Improving the knowledge of, and methods for the economic evaluation of biodiversity and ecosystem services, particularly:**
   2.1. On the environmental, economic and social impacts of changes in ecosystem services;
   2.2. On the contribution of the natural capital and the ecosystem services for the sustainability of economies;
   2.3. Values for the use and non-use of biodiversity and the advantages and limits of the concept of ecosystem services in this regard;
   2.4. The inter-relation of multiple values and their integration in the public decision-making processes;
   2.5. Evaluation techniques for estimation of the cost linked to changes of ecosystem services.

3. **Improving basic political and institutional knowledge, particularly for:**
   3.1. Better understanding of the social, economic and political conditions that underlie policies that have an impact on biodiversity and analysis options for better governance schemes, such as adaptive management or the "ecosystem approach";
   3.2. Evaluating and developing strategies of policy responses and governance structures to preserve biodiversity, taking into account the variability of their effects as a function of ecological and social contexts.

4. **Improving methods and tools for the evaluation of ecosystems, particularly by reinforcing:**
   4.1. The multi-scale approach of the MEA framework in the pan-European context, taking into account the ecosystem services drawn from outside Europe;
   4.2. Data on reference situations, the integration of data, and the production of indicators of ecosystem functions and services;
   4.3. The approaches and methods that take into account the uncertainties, irreversibility, complex dynamics, non-linear changes and participatory multi-scale methods appropriate for the evaluation of biodiversity;
   4.4. Scenarios and other tools for prospective trend analyses.

These research priorities come in particular from the following considerations:

- Through the link that they establish between biodiversity and human well-being, the evaluation of ecosystems for the millennium (MEA) and its concept of ecosystem services, is very important for a new approach to the preservation of natural resources and biodiversity. However, the ecological services approach is complementary to the non-utilitarian rationales of conservation of biodiversity, i.e. based on intrinsic values;
- In order to make several aspects of the MEA concept operational, numerous knowledge gaps have to be filled. To do this, inter and trans-disciplinary research is necessary, coordinated on a European scale, in order to provide specific, sound advice, and on multiple scales of political implementation;
- Institutional changes will be needed, within the scientific community as elsewhere, to build the scientific capacity and the science-society interfaces needed to respond to the challenges of the MEA.

epbrs *European platform for biodiversity research strategy: promoting knowledge for sustainability*

# Appendix VII

# List of tables

# Appendix VIII

# List of figures

# Content