The economic valuation of ecological services

Introduction

This section seeks to present the contribution that the notion of ecological services makes to the socioeconomic valuation of aquatic environments – from the viewpoint of the large-scale water cycle. Many of the arguments that follow are not specific to **aquatic environments however, and have instead a bearing on the management and valuation of the natural environment in general.** One of the difficulties encountered in the economic valuation of services is how it connects with the specifically ecological dimensions of environmental protection. As a result, although this section focuses on the **social dimension of the environmental question,** we will sometimes extend our thought process to include natural environments themselves.

An explanation needs to be given about the swift emergence of the notion of "**ecological" services** in scientific and political debates since the work of the Millennium Ecosystem Assessment (MA) in 2005. It would be tempting to treat this notion in the same way as we have done "sustainable development" which, in the space of a few years, has turned into a sort of semantic ragbag. This would ensure it would become a type of "green" label stuck on everything to do with natural environments and likely to be in our interests – hardly a productive outcome in terms of guiding our discussions on the matter.

We want to show how this notion can serve as a **guide in thinking about the diversely harmonious relations that human societies uphold with their natural environment.** The notion of ecological services is more complex than it first appears, and readers unfamiliar with economic approaches and social sciences in general may at times feel as if they are getting rather bogged down by the arguments that follow. If so, we feel confident that their efforts to persevere will pay off in the end....

- we will begin by putting forward a strict definition of the notion of **ecological services,** halfway between the old categories of "**natural heritage**" and "environmental good" which have long dominated discussions on environmental economics. In particular, we will highlight how closely related this is to the critical question of individual and collective **property rights** on the environment. We will then look at the key steps in the **transition from the natural capital to the services provided by environments.** We will argue that, since there are no "automatic", one-sided correspondences between the status of a natural environment and the level of services provided, an open approach should be taken to this question. The same also applies to the **monetisation of services** in a cost-benefit approach, in which an excessively closed perspective – one which, in a way, looks no further than the purely financial aspects – may lead to serious estimation errors;

- from its purely sociological point of view, the point of valuing ecological services is to link the status of natural environments to diverse categories relating to human "**well-being**", in the broad sense of the term. This is the defining issue of the MA initiative for example. However, for human and social sciences, this "well-being" is not in any way a sort of spontaneous reaction on the part of society to the enjoyment of natural environments – but a **complex historical and cultural** construction stemming from the anthropological and social architecture by which our mediation with nature is organised. Concerning the socioeconomic valuation of services, this mediation architecture can be analysed as a **supply-demand type relationship.** This will be our third main point for discussion in this section. In supply terms, we will focus on the role of infrastructure devoted to **enhancing environments** and therefore to using the services with which they provide society. In demand terms, we will highlight the **importance of individual and collective behaviours** for an economic estimation of environmental well-being;

- throughout, the valuation of services is viewed purely as a tool for finding out and assessing the quality of our relationship with nature – with no underlying political or decision-making goals. That said, since valuation assumes **valuation criteria,** it should for all that be used to form judgements, identify problems and possibly guide the search for solutions to these problems. This **political and decision-making** angle to the valuation will therefore be discussed fourth in this section;

- first of all, we will warn against a sort of naïve "economism" for which the monetisation of costs and benefits would automatically provide the key to constructing a socially and ecologically effective environmental policy. Economic valuation is part of a much larger movement that we will call "**collective environmental responsibility system**" here, a political system introduced by the law and by social practices, and a permanent focus of public debates. Then we will reveal that – in the context of such a system – service valuations need to be talked about in the plural, as public and private stakeholders are called to assess the impact of their actions on natural environments in a variety of ways. These diverse valuation needs will be listed in the scope of environmental action. In this way, we will show that valuation meets two complementary – but partially contradictory – expectations of stakeholders: first, **to provide them with operational decision-making tools** and, secondly, **to shed light in political debates** on the challenges of protecting nature;

- once these expectations have been identified, we will turn our attention to the existing forms of assessment and group them into three categories: **"eco-centred" assessments** conducted in the field of environmental and life sciences; **"socio-centred" assessments** conducted in the field of social sciences; and lastly **"co-assessment"**, an attempt to gain an overview of the interdependencies between societies and natural environments. The early developmental stage of these approaches naturally calls for the emergence of **interdisciplinary methods in environmental sciences,** while at the same time revealing the dearth of **integrated environmental management approaches;**

- our sixth line of thought will broach the more specific problems encountered in **putting an economic value on ecological services.** Part 2 of our report painted a sweeping picture of the knowledge acquired in the large-scale valuation of ecological services. Here, we will focus on **micro-economic approaches,** more likely to be applied at the scale of **local hydrosystems**. There are many methodological guides describing the various economic valuation methods available for the environment, so we won't go back over them – looking instead at what sets these methods on the same footing or apart, and at their **potential for application** in different contexts and at their limits;

- lastly, the seventh part will draw some conclusions and suggest some recommendations as to **putting these methods into practice.**

The notion of ecological services: nature as a unit of stocks and flows

Today, the notion of **ecological service** is tending to replace the categories of "**environmental good**" and "**natural heritage**" that were developed to analyse the interactions between nature and society from the 1970s. While it has the advantage of clearing up some of the difficulties posed by these older categories, it also raises new ones. It explicitly introduces a distinction between the idea of nature as a **capital** – the former notion of natural heritage – and as a space in which environmental goods are produced, or more exactly a flow of environmental goods. We can thus characterise the natural environment by a certain status, defined on the basis of indicators of the environments' functionality – and diversely impacted by human activities. A service supply potential corresponds to this status – in the sense of a flow of environmental goods whose quantity and quality depend on the status of the environment. This saves us the confusions that dogged the previous approaches between heritage-related elements and those relating to the production of goods, or – in other words – between elements of stock compatibility (or environmental balance) and elements of flow compatibility (or of environmental potential use).

The notion of "environmental good" was also fairly loose in these approaches. For example, the possibilities of contemplating river or canyon landscapes (we are thinking of a famous study on the visibility benefits of the Grand Canyon in Colorado), picnicking by a lake or shellfish gathering along the coast were all held to be "goods" (Figure 59) – when there is no actual production of a "good" by the environment in the strict sense in any of these examples. Instead they are illustrative of the supply of a complex recreational service in which water is only one element among others. The notion of "ecological service" thus encompasses both the production of environmental goods in the usual sense (abundance of fish populations or timber for example) and the supply of complex recreational services within a common approach.

Figure 59

a © V. Marty - Onema
b © M. Bramard - Onema

The notion of ecological services includes many different uses of natural environments – from the simple contemplation of landscapes to such practices as shellfish gathering along the coast.

Ecological services, rights to environmental property and use

This notion should also resolve the central problem of **environmental property,** at least in principle. The question of implicit property poses a classic dilemma in the field of environmental valuation. Depending on whether people are asked about their willingness to pay to benefit from an improvement to the environment – or to prevent further damage to it – their answers vary considerably. In the first instance, the person questioned is led to understand that he or she never owned a right to environmental quality in the first place and that this must therefore be bought; while in the second, we imply that this right was initially theirs, but that we intend to take it away and damage the quality of their environment in the process.

Natural assets have been lumbered with **property rights that are either ambiguously sketchy,** or completely non-existent. The notion of services promises a more rigorous legal framework by restoring the distinction – key in Law – between **rights to property and use**. Although the natural capital can be appropriated through land ownership, the value of its services can only be very partially encompassed by the owners: this is the case for private woodland with no enclosure for example. Ecological services, by contrast, go a long way in illustrating the reverse – where the asset (a groundwater reserve for example) is not owned but the services it provides (production of spring water) can be claimed and are thus subject to a commercial use. This brings us to the following typology of appropriation as presented in Table 11.

Table 11 *Types of ownership of services and natural capital.*

		Natural capital	
		owned	**Not owned**
Services	**Owned**	Cultivated agricultural product and land	Maritime fishery
	Not owned	Recreational woodland services	Warming of the West-European Atlantic coasts by the Gulf Stream

Elinor Orstrom's work gives just one example of how, depending on whether or not natural assets and their services are owned, **the quality of their management at individual and collective level will vary significantly.**

Research on ownership statuses has evolved, paying increasing attention to **the use of goods.** In this context, what needs specifying is access to the good or asset. Two types of consideration organise this access: physical ones and social – **legal and cultural** – ones. By physical considerations, we mean **intrinsic properties** to the services: a landscape falls into the general category of public goods in the sense that contemplation of a landscape does not preclude its contemplation by someone else. In this case we speak of non-competitive uses (Figure 60).

Figure 60

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Contemplation, a non-competitive use of landscapes, or the landscape as a "pure" public good.

Regarding production of spring water, we will speak of a **"private" good** in the sense that consumption of this water by one person excludes its consumption by someone else. In the space of services' physical characteristics, we can therefore construct a gradient of conditions for use – ranging from uses that are absolutely not competitive (so-called "pure" public goods) to mutually exclusive uses.

On top of this space we have to add **a space in which access to the services is socially controlled.** The purpose of this social control is to limit access to these services (Figure 61), according to a gradient ranging from prohibition pure and simple to broadly free access. This control can be anonymous – such as totally prohibiting everyone access to a nature spot – but it is usually nominative, authorising different levels of access to different, identified, groups or persons. We often speak of **"club" goods** in this case, the "club" referring to the group of users with access to the service.

Figure 61

a- b- c © L. Mignaux - MEDDTL

Use of natural environments can be subjected to access rules.

The ownership or otherwise of services and natural assets has major consequences for their socioeconomic valuation. Under a **private property system**, it is generally accepted that the owners want to use the assets in their possession in their very best interests. This obviously does not mean that their decisions are necessarily compatible with the public interest, hence the importance of the community keeping tabs on their decisions (problem of "externalities"). It is also clear that the owner may – out of ignorance, negligence or short-sightedness – make the wrong decisions regarding the sustainability of the services potentially provided by the natural asset in their possession, including in their own interests. This justifies an enforcement of standards and regulations by the community as to the **operating practices** implemented by owners (Figure 62).

These reservations aside, it nevertheless appears relevant to identify the value of services provided by a privately owned asset from the benefits it gives the owner – with provision being made for the collective benefits created, which will not generally be owned.

Figure 62

French legislation lays down standards and regulations for operating practices.

How to proceed in the case of services that are not privately owned is less evident, given that their beneficiaries cannot be simply, directly identified. Such services come under **common property** laws in this case, with the underlying asset possibly having a public good dimension too. The traditional axiomatic view of the economy supposes the rationality of decision-makers, focused on maximising their well-being – which justifies **the identification between this maximum well-being and the benefit of the service provided by the ecosystem.** However, such an approach can no longer apply when services are owned by society – except if this society is viewed as a reputedly rational, single individual, which economists generally refuse to do. The major consequence for service valuation is that, to date, the value attributable to services can no longer be dissociated from the **political, economic, cultural and social aspects of these services' management by specific social groups:** rural communities, fishermen's or hunters' associations for example (Figure 63).

Figure 63

a © D. Pujo - Onema
b © A. Fraval - Inra

Environmental management often depends on associations of specific users such as fishermen and hunters.

In many cases it can nevertheless be observed that, even if the underlying asset is not privately owned, its products are – which makes at least a closer valuation of the service possible. This problem of common property explains why the Millennium Ecosystem Assessment (MA) introduced the category of regulating services into their types of ecosystem services. Of the main service categories distinguished by MA (listed below, see section III.4), **supporting** services really do fall within the natural sphere and are neither privately owned nor truly claimable as such by society. **Regulating services** refer to ecosystem functions that may be owned in part – usually under a common property agreement. For all that, the definition of this particular category is rather vague as it tends to confuse social regulation and natural regulation – in the sense of control loops within natural systems.

Valuation of services and sustainability

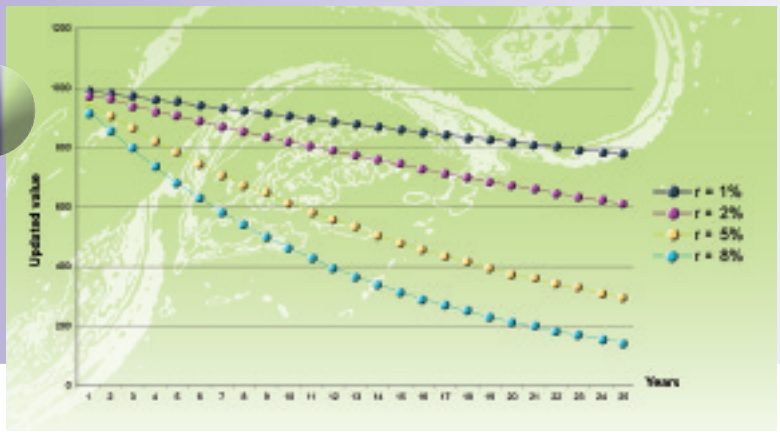
Finally, the distinction between flows and assets introduces the temporal dimension into the valuation. The asset's value as a natural capital should be equal to the **present value of the flow of benefits** associated with the flow of services provided by the natural asset. The idea of present value assumes the introduction of a comparison criterion for the benefits provided at different points in time. It must be possible to measure these against the same yardstick – usually the expected future benefit from the point of view of today. In practice, the calculation is done by introducing a future **discount rate.**

Economists have held countless debates with a view to choosing an appropriate discount rate for measuring the present value of a flow of benefits. A high discount rate reduces the future benefits and therefore the sustainability value of a natural asset (see text box). Automatically attaching less importance to the benefits of future generations (by taking account of compound interest) than for present generations' benefits also brings up sensitive **ethical questions**. This view of the dynamics of service value naturally evokes the concerns of sustainable development, making the valuation of services a key aspect in assessing the sustainability of natural environments and of their functions.

Figure 64 illustrates what a sum of €1000 received in different years in the future would be worth based on the selected discount rate. Rates of 1%, 2%, 5% and 8% are considered.

The 1% rate is used in the Stern Review to assess the future benefits of a global warming prevention policy. The 2% rate is often chosen as the "perpetual annuity" rate and is used to assess the future value of timber harvests in forestry. The 5% rate is recommended by the French Strategic Analysis Centre (CAS) for assessing the future benefits of public projects. The 8% rate is the old discount rate for public projects, applied until 2008 when the CAS revised it downwards to the current 5% rate.

For these different discount rates, the table illustrates the loss of present value of a sum of €1000, over a 25-year period. If we take the CAS' 5% rate for example, we can see that €1000 of environmental benefit collected in 25 years' time amounts to a mere €300 in benefits today. It is this €300 benefit that should be compared to the cost of an environmental protection measure taken today – likely to produce an environmental benefit worth €1000 in 25 years.



In general, substituting the notion of ecological services for the environmental good category shifts attention from the conditions of **an expressed social demand** for the environment to the conditions of a supply of services by natural environments.

This shift should, for all that, be seen as an improvement on previous approaches – which focused above all on demand – rather than as a substitution approach overlooking the importance of demand in favour of supply. This temptation should be resisted, particularly with regard to a development of **ecological engineering**, which is naturally more geared towards the supply of natural services than its conditions of socioeconomic promotion.

From the natural capital to the services provided

Before beginning the central discussion of the links between the **"ecological status" of a natural asset and the type and quality of services it provides** – a key quality of the valuation of ecological services for the purposes of environmental protection – we have a few more thoughts on the notion of ecosystem itself and its limits – which we looked at in Part 2.

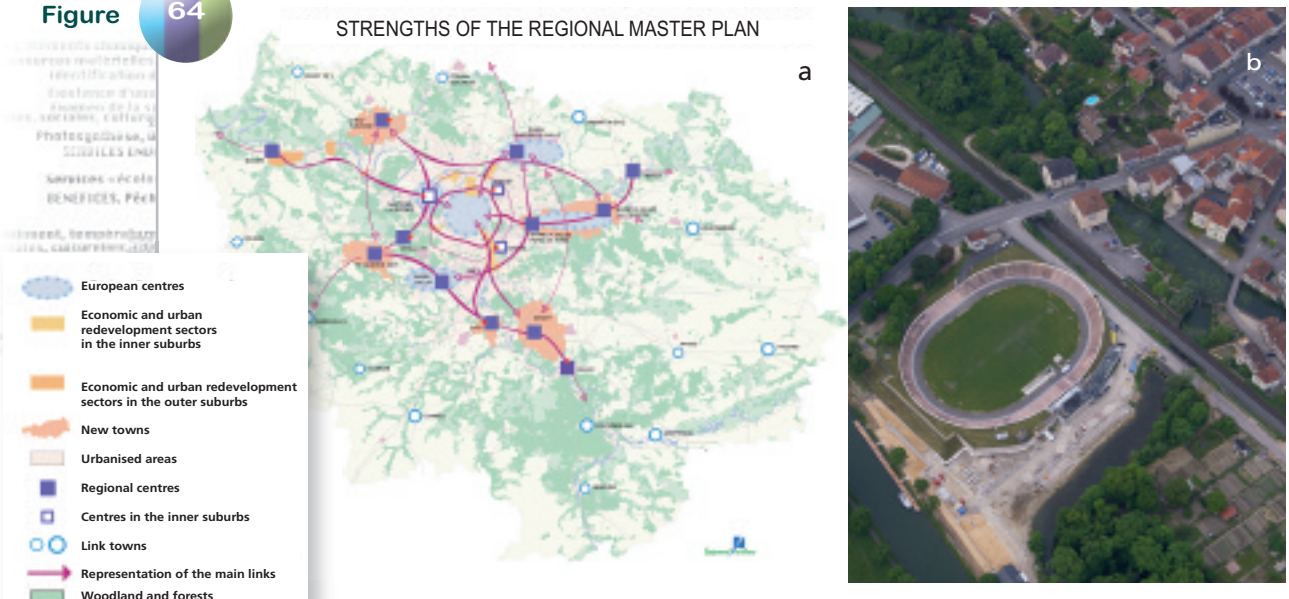
An open approach to the notion of ecosystem

The way we picture environments can be compared to a closed graph of interactive processes. In a nutshell, the "system" dimension would apply first for defining what we understand by ecosystem – in the sense of a **tight network of functional relations between organisms and habitats.** But this narrow-minded "systemic" vision has its faults, since many species interact very little directly inside a particular natural environment. What's more, **organisms can travel from one habitat to another** or meet in diverse types of habitats. In other words, the fact that a particular species is spotted in a habitat does not mean that it belongs to this habitat. Furthermore, relations between organisms are often due to **co-presence.**

These are the usual categories of urban sociology. A city is the result of the processes of co-presence, pooling of available resources on a given site and social interactions all coming together. Habitats (and no longer ecosystems) present similar characteristics to cities, with the proximity of varied organisms or species within a given habitat not necessarily creating a system in the same way that the neighbours in an apartment block can live side-by-side without ever knowing each other – their co-presence in the same place does not create a "system". There are, in fact, models that recognise this dimension of simple cohabitation of species within an ecosystem (Hubbel, 2001).

The urban analogy is also useful for identifying the links between natural capital and ecological services (Figure 64). A city is typically an infrastructure (and therefore a capital) for making services available (habitat, education, health, business, communication, transport) to its habitants. A significant proportion of urban residents' activity is devoted to supporting and developing this infrastructure – either on their own behalf or the community's.

Figure 64



a © DREIF
b © S. Mougenez - Onema

Habitats, and no longer "ecosystems", present similar characteristics to urban areas.

Continuing with this analogy, note that analysis of the continuity value between habitats (creation of **green or blue "corridors"**) follows exactly the same lines of reasoning as the construction of roads between urban areas or the opening up of residential areas (Figure 65). It thus seems worth considering the possibility of going beyond mere analogy and trying to draw genuine parallels between natural habitats and urban areas. Such similarities would make it possible to put the rich potential of urban geography and spatial and geographical economics to use in forming a brand new perspective of natural environment management.

Figure 65

a © P. Mangeot - Onema
b © C. Maître - Inra

The value of ecological continuity follows the same line of reasoning as the construction of roads.

These comments call for an **open approach to grasping the relations** between "natural capital", in the sense of ecosystems present in nature, and the services they provide. It is particularly evident that, in some situations, an environmental management approach involving a detailed description of every single interaction between the living organisms present can prove too complex to put into practice. **With respect to management, "summary"-type analyses,** only looking at the key ecological characteristics of ecosystems, can provide useful guidance to managers as regards the priorities – for example restoring continuity between environments or maintaining their capacities of resilience.

Regulating services at the crossroads between natural capital and services

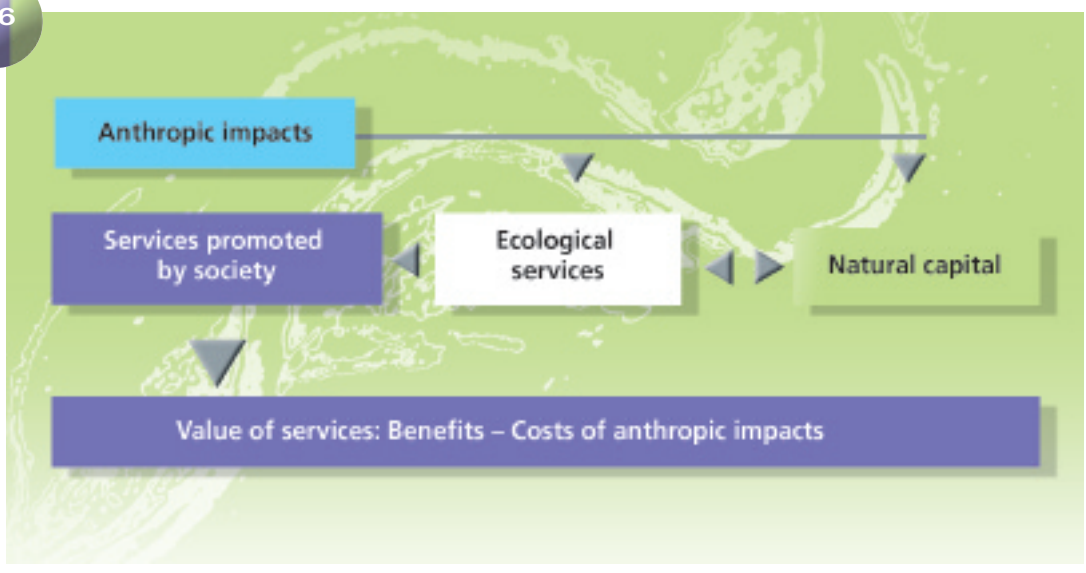
To grasp the link between natural capital and ecological services, we need to mention the so-called "**regulating" services** of the MA typology. Since human activity – agriculture for example – modifies, and even artificially creates (in the case of aquaculture, intensive farming or greenhouse cultivation), the regulating loops of natural processes, this service category should be handled with care. **This is because most natural systems are under some sort of human influence today.** Living organisms are managers of biophysical resources and information on the way to manage them – and also have the ability to self-replicate. "Regulating" services are, in fact, a complex of bio-geo-chemical processes (like large-scale cycles for example) and the results of humans managing their bio-physical environment.

This means that, even if we temporarily remove ourselves from the landscape, **this is still a service category subject to different spatial and temporal scales** and, as a result, we cannot really compare the services to each other. The idea is that, beyond direct interactions between organisms within an ecosystem (competition, predation or parasitism for example), relations within it are forged along **different mediation channels that are partially physical and partially specifically biological.** Part 2 of the report shed extensive light on this aspect and the way in which we can broach the socioeconomic valuation of ecological services as a whole.

Valuation and cost-benefit analysis

Through the valuation process, we also expect to be able to assess the "costs" just as much as the ecological "benefits" of anthropic impacts on natural environments. In this regard, we need to be aware of the possibly limiting nature of the valuation process from the point of view of environmental management. Human impacts bear directly on natural assets and the levels of ecological services, but only indirectly on our uses of these services. A significant proportion of these impacts will therefore in some ways continue to be "borne" by natural environments, without compensation of an equivalent amount in terms of service loss for society. Figure 66 illustrates this problem through a cost-benefit approach to management of natural environments.

Figure 66



Cost-benefit analysis of services and anthropic impacts.

The costs of anthropic impacts are assessed in terms of the **losses of potential benefits** of ecological services given a value by uses. The approach is variational: for a given variation of impacts, the consequences of this variation on users' services are assessed via its effects on the ecological services-capital complex. The economic value associated with this variation serves as the basis for calculating the "cost" of the impacts. **"Internal" effects within the natural sphere are only indirectly factored into the calculation,** in so far as they influence the level of services with which the environments provide us.

The question of long term

This problem can partially be overcome by looking at the **long-term effects of anthropic impacts.** This is because an assessment of the ecological benefits on the basis of variations in service flows only measures short-term effects. To assess the impacts on the natural capital itself, the long-term effects on flows need to be considered. This isn't simply a question of delayed effects, in the sense that the anthropic impact on the flow of services associated with the natural capital is not immediately perceived.

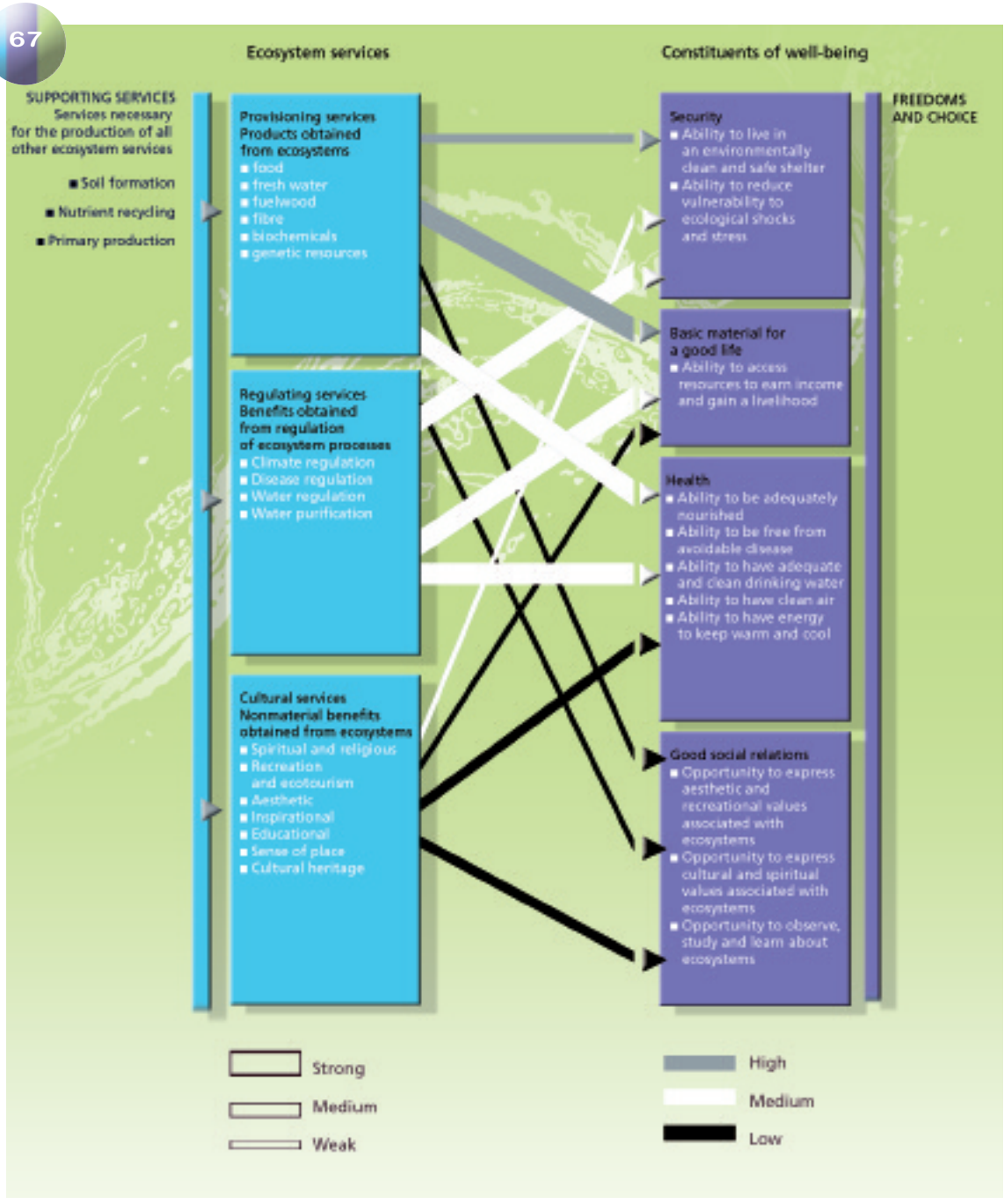
At a deeper level, we have seen that the value of a natural capital is equal to the value of the flow of benefits arising from the services it provides. Correctly assessing the costs of anthropic impacts therefore requires their effects to be considered throughout the period of time in which the natural capital is likely to provide services – typically a long timeframe that can span several centuries.

Managers have a certain tendency to view valuation as a "once-and-for-all" exercise, with a few updates here and there but no real thought as to the actual dynamics of services. If we are to detect "low" signals over the short-term that have much greater consequences over the long-term, instead we must **encourage the creation of permanent observatories for environmental assessment,** capable of monitoring both the dynamics of environments and the anthropic impacts they undergo over the long-term. These ecosystem observatories are as yet few and far between, and we would do well to afford them greater consideration.

Well-being services

The MA's typology is based on a matrix of links between ecological services and socioeconomic categories relating to **"well-being"**, in the broad sense of the term. This well-being is approached from a variety of angles: consumption and use of natural resources, health effects, aesthetic satisfaction, environmental and food safety. The sheer number of different angles makes comparisons and summaries difficult in a global valuation of ecological services (Figure 67).

Figure 67



The benefits obtained from ecosystems and their links to human well-being (according to the World Resources Institute, 2003, and the Millennium Ecosystem Assessment report, 2005).

Mediation between humans and nature as a supply-demand relationship

This brings up one of the major shortcomings of the MA analysis: the oversight of the socially, culturally, economically and technically built architecture organising and regulating **mediation between humans and their natural environment.** There are three main building blocks to this architecture: sée de trois éléments essentiels :

- **a socio-technical architecture** of access to ecological services;
- **an interface of representations** and socio-cultural and socio-psychological connection between humans and nature;
- and lastly **a mediation system** between groups and individuals (Figure 68) through the use of natural environments (markets, institutions, property agreements, and legal systems of environmental responsibility).

Figure 68

Mediation between humans and their natural environment.

To complicate the analysis, this architecture has a history – or rather diverse histories – depending on the socio-cultural area under consideration – and a history that is still being written through ongoing change.

It is important to note that the mediation architecture between human societies and natural environments influences the value of ecological services through all of its three building blocks: socio-technical, socio-cultural and socio-politico-economic. In other words, **ecological services only express a potential of social promotion** – with this potential taking concrete shape through the architecture of mediation between humans and nature. Identifying every single anthropological and social dimension involved in this materialisation would be a step too far. By sticking to what really matters in a **socioeconomic promotion approach to ecological services,** we can say that the mediation interface will produce a local example in the geographical, social and historical space of a **supply-demand type relationship.**

Assessing the value of the ecological service supply

In terms of supply, the degree of social and technical development of human groups will allow for a certain **enhancement** of the services provided by natural environments. This above all consists of **the ability to access the services.** Promoting a natural park assumes roads for getting there and footpaths for exploring it.

Accessing drinking water supposes distribution networks, raw water pumping and purification facilities and wastewater treatment systems (Figure 69). This is what led us, in Part 2, to distinguish the notion of ecological services in the strict sense of the term from that of **benefits** obtained from ecosystems – since the latter includes these different human investments necessary for benefiting from ecological services.

Figure 69

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b © C. Roussel - Onema
c © L. Mignaux - MEDDTL

Access to ecological services supposes suitable infrastructures and developments.

Moreover, we note that these fitting technical interventions can themselves have a positive or negative impact on these services: **enhancement for our sakes very often destroys the worth of protected fragile environments** that it makes accessible – quite apart from any consequences of too many visitors.

Although the question no longer arises for environments that have long been accessible, it is entirely relevant for the development of **protected sites,** since the value of these sites should, at least in principle, be measured in net terms – i.e. by taking account of the loss of benefits that the opening up of such sites to the public will bring about.

Beyond access, enhancement will also entail **different development and conversion operations** aimed at maximising services' potential of socioeconomic promotion. A natural mineral water spring will be equipped with bottling and distribution facilities – and perhaps even spa facilities. Development acts may bear on the **natural capital** itself, such as the creation of spawning grounds, or the provision of resources to the environment: stocking with young fish or release of fauna in hunting reserves. Enhancement for tourism purposes is another example of development vectors structuring the characteristics of environments offered to users. Development can also concern the **flow of services,** either to increase their volume or improve the quality. Forestry and agriculture are two symbolic examples of this. Various points relevant to our discussion are worth noting:

■ From primary to complex secondary services

Enhancement does not just concern a single ecological service. Most of the time it seeks to build a **complex service** by combining and taking specific action on different services. Founding the Landes forest massif (Figure 70) thus led to the creation of a characteristic sustainable landscape capable of protecting the area from dune movements (regulating service) and installed a viable original ecosystem (self-supporting service), likely to be of socioeconomic value (provisioning service). Enhancement therefore brings about the setup of **a supply of "secondary" services** stemming automatically from the "primary" ecological services – which will act as **supporting services.**

Figure 70

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Forest enhancement of the Landes region from the 19th century, an example of human creation of a "natural" ecosystem that provides a wide range of complex ecological services.

On top of their specific ecological impacts, development acts are generally expensive in terms of infrastructure and operating and maintenance costs. Since they are essentially facilities **generally intended to provide multiple services,** in principle it is impossible to attribute these fixed infrastructure and overall maintenance costs to specific services. Consequently, the economic valuation of services will, in fact, measure values derived from secondary services produced by the primary ecological service provision infrastructures, **without explicitly factoring in the costs of these infrastructures.** This represents a serious limit if we want to perform a complete cost-benefit analysis.

Let's look at the underlying paradox. For nature protection associations, the valuation should attribute the "due" value to the benefits that people receive from ecological services. But, for all practical intents and purposes, it is not possible to **distinguish these benefits from the benefits produced** by the development and enhancement operations that enabled them to exist. In other words, between two natural environments of comparable ecological interest – but one of which is accessible and developed and the other not – the former will reveal significant "ecological" benefits where the latter will produce hardly any, leading to the paradoxical conclusion that natural environments need developing if we want to preserve them as producers of ecological benefits.

Furthermore, by its very design – reflecting dated technological possibilities and our flawed knowledge of natural environments – enhancement will impose specific constraints on the methods for accessing and mobilising ecological services. Environmental management is therefore **constantly torn between objectives to protect environments and development objectives** aimed at improving access to and use of the services provided by these environments. Enhancement also has its own dynamics due to technical progress and breakthroughs in scientific knowledge. Last but not least, supply does not ignore demand and developments change as **new needs** are expressed or new demands are made of the services provided by natural environments.

To conclude, **access to ecological services is rarely direct,** but through specific development interfaces. This interface results in the assembly – usually at local level – of diverse primary **services** with a view to producing **developed services** – the actual objects of the social promotion of ecological services.

Demand valuation for ecological services

Now let's look at the way in which demand is expressed for the environment. We have already mentioned the fact that most environmental valuation studies have traditionally focused on the demand of environmental goods. We now need to be more precise in defining this approach.

From impact measurement to behavioural analysis

Demand studies themselves followed on from even older approaches inspired by **dose-response models** that are common in ecotoxicology and epidemiology. These were based on the notion of **"impact function",** here the impact on people of the biological and physico-chemical characteristics of their environment. Costs in terms of morbidity or health risks can be associated with these impacts.

Although they don't completely reject the merits of such approaches, human and social sciences seriously limit their scope on the grounds that they don't take sufficient account of **individual or collective behaviour** in the face of environmental risks. These sciences will typically prefer approaches explaining the role of behaviours in risk exposure over the former – and more generally in studies of the relationships between humans and their environment (Figure 71).

Figure 71

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Risk management above all involves understanding and acting on individual or collective attitudes and behaviours in the light of risks.

As for more strictly economic approaches, these assume that behaviours express the merits (or "value") of environmental characteristics for individuals or societies. This is the principle of "revealed preference", here revealed by the consumption (or non-consumption) behaviours of environmental goods and services.

The first strength of this approach lies in its potential for application. Since individuals in this regard are supposed to want to get maximum benefit in terms of well-being out of their possibilities for accessing ecological services, the strategies they will implement to this end tell us what value they attach to the environments. In general, it is very difficult – and borderline impossible – to directly measure the well-being that society derives from natural environments. It is, however, possible to observe behaviours (visitor numbers, leisure activities, economic exploitation for example) and thus to indirectly deduce the well-being created – which may then be related to the services creating it.

The second advantage of this approach with regard to methods based on impact functions is that it does not implicitly assume that the human-nature relationship casts an "average" person with regard to biomedical or social constants and an "ambient" or "average" environment.

Behaviours vis-à-vis nature are eminently variable from one person to the next, reflecting the variability of environments as well as the attitudes and preferences of individuals for the environment or their specific socioeconomic characteristics:

- level of income;
- education;
- sex;
- age;
- lifestyle.

In other words, the environmental demand valuation approach does not merely make do with estimating an "average" well-being, but seeks to construct a distribution of well-being, of which the variance analysis is particularly enlightening as to the relationship between people in society and their environment.

The role of behavioural approaches in service valuation

These comments show that the environmental valuation approach based on demand will in fact involve a **study of individual and social behaviours.** Behavioural approaches are currently being developed significantly in economics just as in other human and social sciences. They are tending to go beyond the traditional paradigm of revealed preference, which has turned out to be rather limited, by introducing such new notions as the "framing" of preferences.

"Stockholm" Syndrome is one such well-known example, where a hostage becomes sympathetic towards his or her captor. In a context of extreme psychological stress, this paradoxical attraction is a resistance option offered to the victim, enabling him or her to mentally turn an unpleasant situation into a pleasant one. Smokers' attitude to their addiction is another well-known example of this. These are key points in the contemporary analysis of attitudes to risk and of **precautionary behaviour** in particular. The idea is that **behaviours are not neutral regarding the well-being they seek.** Instead, they exert a framing action on preferences and influence the cognitive content of dilemmas faced by individuals.

This type of phenomenon has little bearing in the context of simple ecological services, such as the value attributed to high-quality raw water for bathing or to a lakeside landscape for example. However, if the valuation exercise concerns the creation of floodplains or introduction of a reserved flow policy aimed at preventing the collapse of fish populations during droughts, the social and individual acceptability of such actions will depend on the way in which the stakeholders – citizens, residents and users of the resource – mentally construct the problem and grasp the environmental stakes attached. Behavioural approaches really come into their own in such contexts.

The contamination of raw water, and even more so the problems raised by what should be called "new" ontaminants (Figure 72) – medicinal residues, organometallic compounds – is undoubtedly one of the priority fields for the application of behavioural approaches. More than their behaviours in choosing whether to drink tap water or mineral water, it is the opinion of individuals as to the importance of the problem that matters here.

Figure 72

© M. Carrouée - Onema

Medicinal residues in water raise the question of the public's perception of the danger a nd its influence on political decision-making – beyond the objective risks of these contaminants.

The way in which individuals perceive the danger posed by these contaminants in turn exerts pressure on political decision-makers. It can lead to the latter taking very costly action to reduce risks that may be very low – to the detriment of other actions more profitable to the natural environment, but wrongly perceived by the public as being less urgent. This type of situation throws up a key research question for social sciences, alongside ecotoxicological approaches seeking to assess the objective risks posed by these contaminants for human health or the status of natural environments. This raises a familiar point in the debate about implementing the precautionary principle.

From demand to the value of ecological services

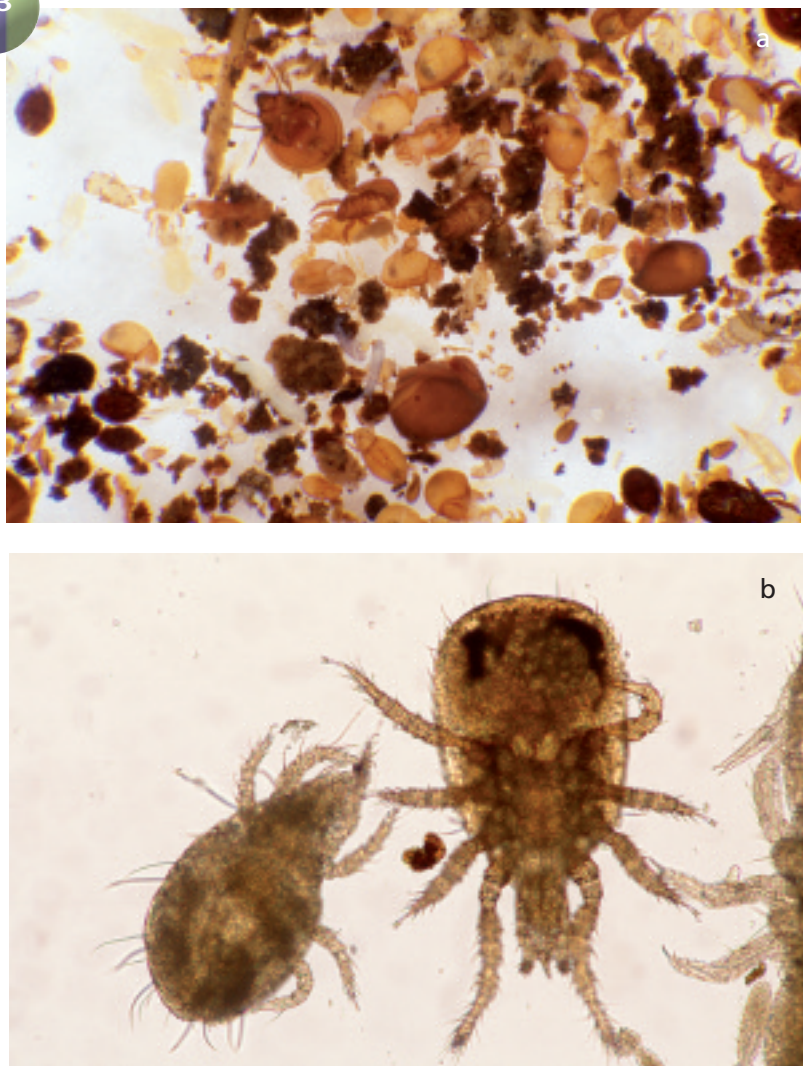
Taking as read that if we talk of demand, we talk of demand behaviours addressed to the environment, let's now specify the way in which the expression of these demands will influence the value attached to ecological services. This expression is made through diverse socio-technical facilities: going to wetlands supposes that a means of transport is available – generally a car; practising water sports requires suitable equipment. Two important points need mentioning here:

- first of all, **demand influences the supply of services,** and therefore the complex of primary services that will actually be valued socially. Since demand is mobile – subjected to prevailing phenomena and dependent on the time available for leisure or level of income – we might say the same concerning supply on the need for follow-up over time of the values attributed to services if we want to obtain a measurement that is at least a little bit stable of the value on the basis of secondary services;
- secondly, demand wields considerable influence over the extent to which services are mobilised, but **it can also affect the very existence of these services.** This is the difficulty in identifying and measuring **"existence" values** attributable to ecological services. Economists have been grappling with this issue for some twenty years already. Although they agree on their importance in a valuation exercise, they differ considerably over the manner in which they should be considered and the status they should have in a valuation approach – alongside more traditional measures associated with the actual use of environments.

Collective environmental responsibility systems

At a deeper level, the notion of existence value takes us beyond the traditional utilitarian paradigm of economists to look more closely at the notion of "values". Without going into the important aspects for human sciences of this notion, we will say that it raises **the question of politics in a valuation context** – politics understood in the broad sense here of the collective implementation of a town's affairs. The expression of private demands for the environment encompasses but a very small part of the scope of environmental values. There are some key ecological services for the sustainability of environments that are not subject to any solvent demand (Figure 73).

Figure 73



a-b © M. Fouchard – Inra

The activity of soil fauna is vital, but is not subject to any demand from society.

What's more, the environmental impacts of human activity are negative external effects that need managing at collective level when incorrectly addressed by individuals.

Rather than positing "social" demand hypotheses for ecological services, **it is more relevant to address the issue from the point of view of the stakes for the community of protecting environments.** The expression of value systems of social groups in the political arena takes material shape as a matrix of targets, resources and constraints for collective action. In the same way as a whole is not always the sum of its parts, the value attached to the environment by a community cannot be broken down to the sum of the values that each of its members attaches to its protection. This collective value can involve future generations – yet to be born – and can also correspond **to the expression of a collective awareness of our environmental responsibility.**

In any society, many standards and regulations cannot be justified by a calculation weighing up their costs and advantages at the level of each individual and then added together. In this sense, the value of the collective action (Figure 72) is partly independent in the utilitarian calculation field that inspires most of the economic environmental valuation methods.

Figure 74

a- b © L. Mignaux - MEDDTL

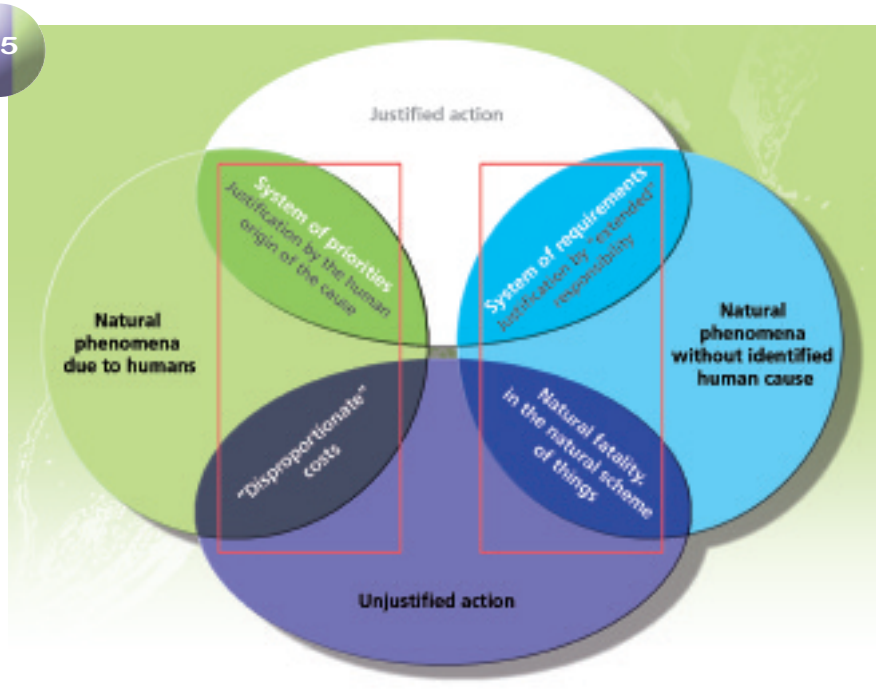
The social value of the action in favour of natural environments does not merely equal its immediate benefits for each individual taken in isolation, rather it includes the benefits for the community as a whole and for future generations, yet to be born.

Planning for collective action cannot completely ignore **the monetisation sphere** for all that: public policies carry both a direct and indirect cost in terms of constraints affecting the satisfaction of individual desires, via the

law and taxes, or of making economic activity profitable. They also have **collective benefits,** but their attribution between individuals is not generally possible in this case. Consequently, the cost-benefit analysis of public action cannot be the sole justification or invalidation of this action, and must instead be designed as an important – but not the only – guide in making our collective decisions. This relativity or relative value of valuation should not lead to the valuation scope being restricted simply to what can be immediately measured in terms of the socioeconomic consequences of public action. Instead, it should be expanded to consider the **social value of standards and regulations** that will organise the collective action in favour of natural environments.

These standards, laws and regulations are, in fact, the concrete reflection of a certain **collective environmental responsibility** system. This system is based on a **responsibility doctrine** founded on two distinctions: a distinction between all of the natural phenomena that can be attributed to human action and those for which it is not responsible on the one hand, and a distinction between a series of natural phenomena necessitating human correction – whether their causes are anthropic or not – and a series of natural phenomena that does not justify action, either because humans are not responsible or because the costs outweigh the benefits that the community can hope to obtain from its efforts to act on them. Figure 75 illustrates the construction of an environmental responsibility system.

Figure 75



The construction of an environmental responsibility system.

Justification for human corrective intervention may be grounded in the human cause of the phenomenon, or a principle of "wider" responsibility (humans as "guardians" of nature for example). Justification for non-action, in the case of phenomena without human cause, is found in the idea of fatality or the nature of things. When humans are responsible for the phenomena, non-action will need to be justified through a **cost-benefit** type argument. Note that this is the only instance in which cost-benefit analysis is called on in general.

An environmental responsibility system is traditionally constructed, which means it changes constantly as societies change and new facts come to light. In this regard, the transition of its line of reasoning from natural fatality to phenomena influenced by humans marks the major development of the environmental responsibility system at the end of the last century. Lastly, it fits into a given socio-cultural and jurisdictional area.

The action/non-action gradient is organised differently depending on the causes of the phenomena in question. In the case of phenomena for which humans are responsible, the action is generally organised according to

a **principle of priorities.** When humans are not to blame, it is **a principle of requirement** which justifies action or inaction. In the former case, the debate on the priorities will call on the valuation approach, while the argument of requirements applied to phenomena considered to be "non-human" falls outside of this area – at least in principle.

Values and value judgements

This is one of the challenging aspects of defining a policy for managing natural environments. Once it has been understood that socioeconomic valuation – or more generally common social attitudes towards natural environments – should not form the only guide for environmental management, we are faced with the tough question of what **other criteria** need to be considered. Political decision-making involves making choices within the confines of available means. This may therefore entail "sacrificing" some environments in favour of others, considered to be more "important". While socioeconomic valuation may not be the chosen criterion for making this decision, it is likely that scientists will have to formulate opinions on such priorities – i.e. pass value judgements on the environments worth protecting. Scientists' reluctance to make **value judgements** is well-known, as it is one of the pillars of their professional ethics and, more generally, their way of seeing the world. This is particularly true in the field of ecology.

This problem is of the utmost relevance to so-called "compensation" policies. According to this principle, degradation of a natural environment can be tolerated as long as a protection operation is undertaken on another natural environment aimed at "compensating" the ecological loss suffered (Figure 76). Judging the value of such a compensation supposes comparing what is lost with what is gained, and therefore implicitly admits the existence of scales of equivalence, and therefore values, applying to natural environments.

Figure 76

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Restored wetland. "Compensation" development operations bring up the question of comparison terms between the ecological value of what is lost and what is restored.

Two responses can be perceived to this problem. The low-level response, and the most realistic in practice, would be that scientists present a **rough classification of the protection priorities,** by leaving it up to the political decision-makers to take their pick from among these priorities. While not necessarily being very satisfactory, the result will nevertheless allow science to stay above the debate and avoid having to deal with the **problem of values.** The high-level response, suggested by Norgaard along with others (Norgaard, 2006), would involve the opposite: scientists facing up to the problem of values in scientific terms, i.e. by taking a theoretical approach to thinking about **environmental ethics.** This idea is reminiscent of Amartya Sen and his plea for a proper science of social and economic ethics. Such an approach would mean rewriting the current role science plays in our societies and expanding our view to include a consideration of the way collective responsibility systems are evolving in our modern societies – beyond the simple environmental debate.

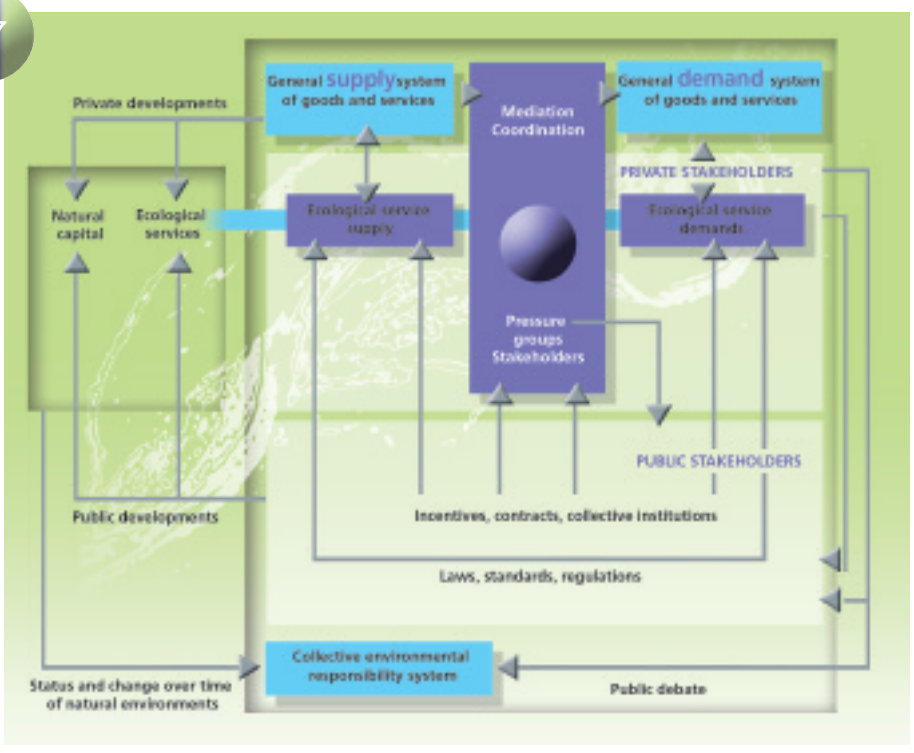
The scope for ecological service valuation

It is now possible to locate the valuation scope with regard to the private and public action scope. Private or public stakeholders have various action means at their disposal, falling under the following categories:

- development;
- exploitation of resources;
- standards, regulations, labels and charters;
- contracts;
- financial incentives.

Although the legal capacities of private and public stakeholders naturally vary considerably, they both have access **to all of these drivers for action,** at least in a constitutional state. A certain intellectual laziness often makes us forget that private stakeholders are, de facto or by right, institutions and as such have a certain capacity for enacting rules and standards – whether for households or businesses. As a result, both private and public stakeholders have to deal with the problems of assessing their actions – needs that can arise directly or indirectly from valuations of ecological services. Access to the right information is key in a public valuation approach, and naturally concerns, in the first place, the difficulty of obtaining private information – but also of putting to use the knowledge gathered in local, and even national, public services.
Figure 77 relates these concerns to the ecological services and decision-making processes involved.

Figure 77



The considerations when deciding on a valuation of ecological services for private stakeholders and public authorities.

To avoid overloading this diagram with even more detail, we have omitted to represent private decision-makers' acts on drawing up contracts between them or on **voluntary procedures** (eco-certification, eco labels for example). The diagram illustrates three important facts.

Assessment in an operational context

Whether to choose between several possible actions, determine the dimensions of an action or justify it, private and public stakeholders will to some extent assess their projects or the situation they are facing. To this end, they will perform assessments as necessary of the ecological services affected or mobilised by their action. When wondering about a general approach to assessing services, we should not overlook the fact that, **in reality,** we need to speak of **service "assessments" in the plural** – which go into more or less detail and generally focus on specific aspects of the value of certain services.

This is a prerequisite in all operational assessment approaches. Private and public stakeholders will only commit to the approach in so far as it bears on the decisions they have to take and the problems they have to manage. One of the main reasons for the current lack of environmental studies is that they are **at odds with stakeholders' operational expectations.** Contrary to what we often hear, the fact that these studies have for the most part been confined to the research field of environmental sciences is not due to a lack of methods or reliability of existing methods, but the difficulty encountered in translating their findings into operational conclusions.

The same can be said for the WFD's reporting activity in Europe. This reporting is committed more to conforming to European requirements than to giving consideration to how the information produced can be used in setting up programmes of measures. Until we manage to tie the information produced on environmental assessments together with the design and implementation of actions at the local level, environmental protection stakeholders will continue to encounter difficulties with usefully mobilising this information.

Assessment and context for action

The table below also illustrates the fact that there are three main dimensions to environmental action: the development and exploitation of environments, the regulation of actions and behaviours, and lastly the financing of action or compensation of damages. Different assessment needs correspond to these different dimensions – Table 12 summarises them.

Table 12 *Assessment needs depending on the context for action.*

Action	Types of assessment required
Development, exploitation	Cost-benefit analysis of the development action Exploitation profitability Analysis of the impact of developments or exploitation practices Measurement of the negative or positive effects on ecological services
Regulation	Effects on the behaviour of stakeholders concerned by the regulation Costs of the efforts that stakeholders need to make to conform to the regulation Choice of the best regulation (comparative performance measurement) Dispute assessment Assessment of damages and compensation (e.g. in a declaration of public utility [DUP]) Assessment of the cost for enforcing the regulation Assessment of the cost of the monitoring to be set up Calculation of fines and sanctions if the regulation is not complied with
Financing	Comparative assessment of the possible means of funding the action Assessment of the financial dimensions and payment schedule Assessment of the stakeholders' ability to pay Assessment of the redistributive impact and compensation needs Damage assessment

Table 12 illustrates the variety of contexts in which an ecological service assessment can be carried out – to provide either direct guidance in decision-making, or information or justifications for a decision.

The twofold role of assessment

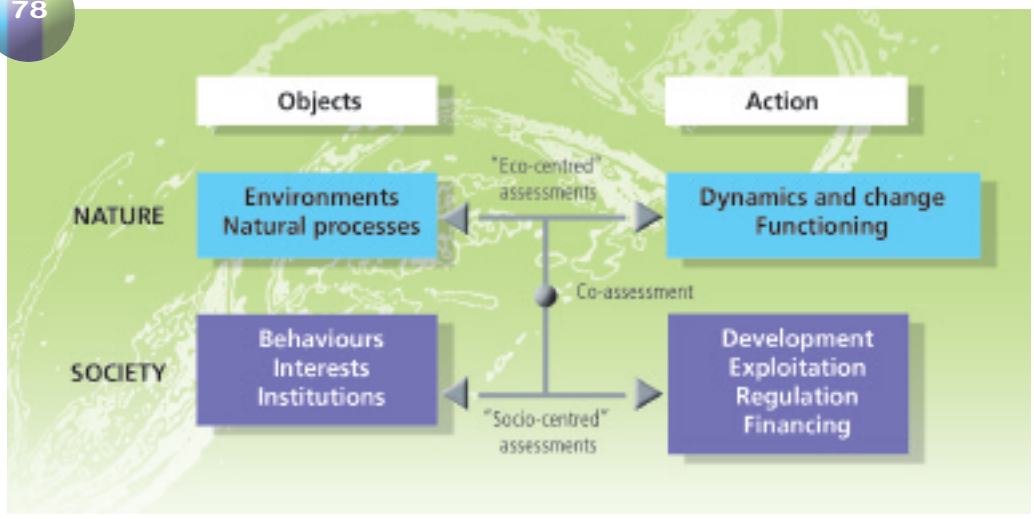
In addition to this purely decisional dimension, assessment also helps to improve our scientific knowledge of environments and provides food for thought in the public debate on the definition of a collective environmental responsibility system. This contribution then feeds back to the decision-making stage, influencing the fields of development, regulation and financing. As we can see then, service assessment meets both **a social need for information to guide public and private action** on the one hand, and the **operational needs of public and private stakeholders within the framework defined by these guidelines.**

One of the major difficulties of assessment is to combine both these dimensions. Operational needs typically concern specific ecological services at the local level, while the public debate focuses more on ecological services in the broad sense – at a national or European level as far as we are concerned. This requires a methodological coherence that is tricky to achieve in practice. For this, we need to give ourselves the appropriate study means in terms of meta-analyses, transfer and aggregation of values at different spatial scales.

Assessment types

Now we need to describe the **types of assessment** that are likely to meet both the operational and informational needs of the public debate and political decision-making. As we have seen, the point of assessment is to bring " objects" – whether natural or social – under the same microscope as stakeholders and action considerations. Figure 78 illustrates this process.

Figure 78



Types of ecological service assessment

Figure 78 reminds us that all living organisms are also stakeholders that must make decisions about how to act. It distinguishes three types of assessment:

- **"eco-centred" assessment.** Since human action is understood as a forcing variable (or external "environment" variable if you like), the assessment focuses either on the impact this action has on environments or on the specific dynamics and functions of these environments, without referring primarily to human action – viewed as one impact among many. This type of assessment leads to measurements of ecological service supply and natural conditions affecting both their volume and quality, as well as an appraisal of these services' sustainability conditions;
- **"socio-centred" assessment.** Since environmental functioning is treated as an external forcing factor, developments, regulation and financing need to be combined as effectively as possible to meet the environmental objectives of a collective environmental responsibility system. The purpose of this type of assessment is to measure both the advantages and social costs of implementing these objectives. It also supports the necessary institutional updates of decision and deliberation plans and measures the environmental performances of public action and private initiatives;
- **"co-assessment".** This straddles natural and social systems. Unlike the previous two types of assessment which aim to associate objects with action or activity considerations, co-assessment seeks to compare the actual assessment methods themselves – whether these concern assessments performed in the field of environmental sciences or that of human and social sciences. In that sense, it assesses the assessment in some ways. Moreover, it informs eco- and socio-centred assessments with the aim of thinking outside their box. A further purpose is to identify the weak – but potentially important – signs of future repercussions, in the social and environmental fields. Lastly, it must be able to produce a prospective – and even predictive – assessment, as soon as enough is known about the processes in play.

Economic valuation of services

Micro- or macroeconomic valuations

It is common to hear that the main difficulty posed by ecological service assessment is methodological. In fact, by "methodological" problems, people generally mean that they are having difficulty pinning down what they intend to measure and for what purpose. **Assessment involves reading a situation or a development on the basis of a set of assessment criteria.** These criteria naturally differ depending on the decisional or informational objectives being pursued. As mentioned above, only eco-centred and socio-centred assessments meet the usual models of an assessment exercise. Co-assessment, which seeks to set overall criteria for different assessment systems raises specific problems of its own that we will not go into here.

It should be clear by now that economic valuation is an essential part of any assessment exercise of ecological services and environmental policies. **It is also clear that economic valuation only addresses one aspect of the assessment problem – alongside assessments produced in the field of environmental and life sciences and the field of other human and social sciences:** anthropological, sociological, historical, socio-political or legal analyses of the problem of ecological service supply and management. These reservations aside, economic valuations are typically conducted at two levels that need clearly setting apart.

■ "Macro" valuations

The valuations performed by the MA or discussed in Part 2 of our report are typically "macro" valuations. They involve **constructing, by aggregation, a global measure of the economic value of diverse ecological services –** usually, in fact, of diverse natural environments at country, continent and even worldwide level. The point of such valuations is not to steer public decision-making in any particular direction, but to provide **information about the value** that our societies need to attach to ecological services if they consider nature to be a capital – in the same way that they attribute a value to the artificial capital they create in industry, agriculture, the financial sector or real estate. To this type of measurement we can add the work conducted on **"green" national accounting** aimed at completing the usual construction of the commercial gross domestic product with an assessment of the contribution natural environments make to the wealth of nations.

One of the major difficulties of these studies stems from their construction on the basis of small-scale micro-economic studies. The aggregation of partial figures does not simply boil down to an addition – due to the high risks of double counting. More generally, the construction of reliable measurements at a global level calls for the **possible substitutions between environments and services** to be considered. An aggregation by simple summation of local values implies that all environments are "specific", in the sense that they are subject to specific and mutually exclusive demands. This specificity of course has nothing to do with ecological specificity, but concerns social uses.

As highlighted in Part 2, the loss of a mineral water spring may be serious problem at local level – in terms of environmental quality as well as jobs or income for residents. At a national level, the effects of this loss will not be felt as mineral water consumers can easily find substitutes for the spring in question. As a rule, the choice of an aggregation criterion supposes beginning with the final scale intended (national, European or worldwide) and working from the demand for services as perceived at the scale selected. The possible substitutions then need to be examined to identify the contribution of a local environment to the whole. Aggregation must then take place by weighted summation according to the imperfection of the substitutions between environments.

■ "Micro" valuations

At the "micro" level, the assessment of services is perceived as **a tool both for informing the public about environmental values and for supporting public decision-making.** Addressing specific natural environments or services at local levels, "micro" valuations seek to reveal the demand for ecological services and the economic conditions presiding over their provision to users – what we have called "the supply" of services by environments – which we have seen is based on dedicated artificial infrastructures.

Part 2 of this report goes into the subject of macro-valuations in ample detail. We will thus focus more on micro-valuations here, which are more in line with the immediate needs of managers in the context of the WFD. We adopt the decisional framework of the hydrosystem as described in Part 2. **We therefore assume that a relevant spatial unit has been defined upstream for the management of services – from the point of view of ecology and the management unit of the territory.**

The principles of economic valuation

As we have mentioned above, economic valuation entails a **study of individual and collective behaviours.** Such behaviours are supposed to reveal certain attitudes towards the natural environment. In economic terms, an attitude is expressed through actions: visiting natural environments or not, practising certain activities in them or not, politically campaigning for the conservation of natural environments or not. For an economist, an action has two key characteristics: on the one hand, **it is the result of a choice** from several other possible actions; on the other, **it mobilises resources,** time and energy, which are all factors that have a cost for the stakeholder.

Choice means preference for one action over another, and cost means that the stakeholder will only act if he or she considers it worthwhile. It is this "worth" that defines the "benefit" of the action, or its "value". That said, diverse "benefits" can be associated with different actions possible. The preference principle therefore stipulates that the action ultimately picked should be the one that corresponds to the highest benefit out of all the possible actions. In other words, the economic valuation of services tries to pin down the value – from the benefits of the chosen actions – that can be attached to their material provider.

Once both these principles have been grasped, all so-called micro-economic valuation methods of the environment are simply specific versions thereof.

Take the simplest such example, that of measuring **defensive or protection expenditure.** Let's assume that we want to measure the economic value attached to good quality drinking water. The consumer has two possible choices: to drink tap water or mineral water. These two options will not cost the consumer the same. The preference principle stipulates that, if we observe someone who has decided to drink mineral water, it is because he or she considers the extra cost incurred to be less than the benefit obtained in terms of water quality. Vice-versa, consumers who opt for tap water must think that the extra cost isn't worth it – when weighed against the difference in quality between tap water and bottled mineral water.

What is interesting in this calculation is the **behaviour switch limit,** i.e. the cost and quality difference that causes the consumer's behaviour to veer towards one option or the other. At this point of indifference, the consumer should quite simply not care whether he or she drinks tap water or mineral water. The text box below illustrates how an economic value can be attributed to an improvement in the quality of drinking water on the basis of consumer behaviour and the difference between the cost of bottled water and tap water (Text box and Figure 79).

A valuation example: choosing between mineral water and tap water

VT is the value attributed to tap water, VM the value attributed to mineral water and CT and CM are the corresponding costs. Typically, we expect CT<CM. Drinking tap water is less expensive than mineral water. At the point of indifference, we should see:

VT-CT = VM-CM

Such that: VM-VT = CM-CT. The difference in value attributed to the quality of drinking water lies in the extra cost. Two comments need to be made at this point.

The values attributed to water quality are overwhelmingly subjective and impossible to observe directly. However, the cost difference is objective, and can be observed. Logically, at the point of indifference, we simply need to measure the cost difference to deduce the difference in value attributed to the underlying water quality. Secondly, it is clear that VM-VT > CM-CT for mineral water consumers and that the reverse is true for tap water consumers.

Let's now suppose that we want to improve the quality of tap water. So VT'>VT is the value attributed to this water after improvement. Let's suppose that the price of tap water stays the same – so the cost difference does too. But now some people who used to prefer drinking mineral water will change their minds and start drinking tap water. These are all those for whom: VM –VT > CM-CT > VM –VT'. Since they save CM-CT and, by noting N the number of individuals changing their minds, the gain for the whole population of the improvement in tap water quality becomes (CM-CT)xN. It is this gain that measures the monetary equivalent of improving the quality of tap water.

Figure 79

a © B. Nicolas - Inra
b © C. Roussel - Onema

The decision to drink mineral or tap water is based on the economic value that consumers attach to the quality of drinking water.

An equivalent way of thinking would be to reduce the price of tap water or hypothetically increase that of mineral water to measure – by difference – the value consumers attach to the current quality of water.

Methods for the economic valuation of the environment

Despite their differences, the other commonly used valuation methods follow the same approach. They fall into two categories: so-called **"indirect"** methods, which deduce the value of an environmental good from the market context in which it is used; and so-called "direct" methods, such as **contingent valuation,** aimed at directly getting each person to reveal the monetary value he or she attaches to the environmental good. But this difference only concerns the survey methods used and not the underlying behavioural model they postulate (Table 13).

Table 13 *Methods for revealing environmental values*

Indirect methods	Direct methods
Protection expenditure	Contingent valuation
Travel cost method	Contingent classification
Hedonic pricing method	

Indirect methods comprise **protection expenditure methods,** which we have just mentioned, travel **cost methods** and **hedonic pricing methods.** The travel cost method works from a measurement of the expenses an individual pays to go and stay in a nature spot of interest. The idea is that the individual is all the more prepared to part with significant sums to visit a nature spot when he or she considers it to be of high value. Different sites can therefore be ranked in order of the expenses devoted to visiting them. The same method is applied to the choice between tap water and mineral water.

The hedonic pricing method involves observing the differences between property prices depending on the property location and the quality of its natural environment in particular. If all the other property characteristics are the same, we expect higher prices for properties located in a protected natural environment than for those situated in a damaged environment. It is this difference between the prices of properties – comparable in all other respects – that will tell us the quality value of their natural environment. Once again, this follows the same lines as presented above.

Lastly, **the contingent method** involves asking people directly about what monetary value they attribute to the environment in terms of willingness to pay for its protection or improvement. But their virtual choices will now be based on the amount of this monetary sacrifice: either lose the well-being they would get from the consumption corresponding to this amount, but gain an improvement in environmental well-being, or keep this well-being but lose out in environmental quality. The underlying behavioural logic is the same as in the previous examples.
The methods we have just described are mainly employed for users of natural environments; however, it is perfectly possible for the economic thinking behind them to be applied to farmers or manufacturers.

For a farmer using irrigation for example (Figure 80), **the value of the water will be the same as the extra profit** that he would gain from using it. If we set a price for irrigation water, the farmer will consume water until such time as the extra profit that he would gain from the consumption of extra water would be equal to its price. Beyond this point, the farmer would lose money and, up to it, he could earn more by irrigating more. The only difference with the previous methods is that the problem of subjective value attribution is not an issue here. An economist armed with all the necessary data could, in principle, reconstruct the farmer's thought process and come to the same conclusions as the farmer concerning irrigation of his farm.

Figure 80

© M. Bramard - Onema

The economic value of water can be subjective, as in the case of drinking water, or objective, as in the calculation of the agricultural value of water for irrigation users on the basis of its profitability for crops.

■ The problem of created benefits

One of the difficulties that valuation practitioners often encounter is how to assess **"created" benefits**. For example, protecting an aquatic environment will lead to a boost in tourist numbers – thereby creating jobs in local restaurants and hotels or reviving local trade in the tourist season. Managers – and even more so local councillors – are obviously tempted to include such created benefits when assessing the consequences of their environmental initiative. But it is clear that, with no structural developments, tourists will go elsewhere, with no new gain for the national community. The problem of substitute sites has been described in the presentation of "macro" valuations.

On top of the fact that these created benefits are generally very difficult to measure correctly, **they should not be taken into account** – what matters are the values associated with the actual environmental action. In concrete terms, if enhancement of a river makes it more popular to fishermen, it is the well-being offered by the extra days' fishing that should be measured, and not the increase in catering revenues from the fishermen (this isn't the same well-being). In practice, it is often difficult to resist the temptation of adding the created benefits to the actual benefits obtained. In this case, **it is preferable to present these two categories of benefits separately.** Only the latter should be counted as "environmental" benefits, since created benefits fall under the category of impacts stemming from action for local residents (local boost in well-being, which is not specifically environmental).

■ Difficulties and limits of economic valuation

Beyond their specific methodological difficulties, we can see that the relevance of the findings obtained by these valuations depends closely on the following.

First of all, **the options from which stakeholders can choose must be known in advance** and specified as far as possible. Since measurements are mainly carried out on the basis of behavioural variations that are observed or simulated when we vary the environmental context, any omission of possible choices or errors of assessment on the underlying motivations for behaviour will result in significant measurement errors.
But stakeholders themselves can have no more than **a confused perception of the choices** they have and of their positive or negative consequences for them – which will lead to assessment errors in the analysis both for indirect methods based on behaviours observed and direct methods, contrary to the claims of the persisting criticism surrounding contingent approaches.

The gap between subjective perceptions and reality is to blame here. Beyond a naïve methodological

cynicism for which the correct measurement of an error of judgement describes the truth because it is correct, the problem of distorted perceptions raises the general question of what information the stakeholders questioned are given. In the context of the large-scale water cycle, public information is of the utmost importance (Figure 81). We can see that it contributes indirectly to improving the quality of socioeconomic valuations by reducing users' assessment errors.

Figure 81

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b © L. Mignaux - MEDDTL

Public information is vital for reducing the gap between subjective perception and the reality of the situation of natural environments.

Environmental valuation practitioners are very much aware of this problem now and most of the latest research addresses it. Conventional approaches by opinion surveys among the general public are now increasingly completed systematically by experimental economic studies in laboratories.

Such experiments bring together a small number of subjects who are given as much information as possible on the choice they are going to have to make. Drawn up in liaison with psychologists, the details of how the questionnaire will be presented, the vocabulary used and the implicit information unsaid will be scrutinised as cognitive distortion factors of the answers. The laboratory work also makes it possible to consider actual payments of individuals from an initial budget granted at the beginning of the research. The link between the decisional problems on environmental protection and choices made under uncertain circumstances also marks the progress made in the relevance of protocols and the analysis of their findings. Experimental approaches show great promise for the future and should soon feature among the usual range of tools available to environmental valuation practitioners.

"Opportunity cost" approach

Opportunity cost approaches may be well worth adopting to meet the specific valuation needs of both ecological services and the protection policies drawn up for these services. The notion of opportunity cost fits into the same framework of economic choice described above. On first appearance, the opportunity cost is what the decision-maker renounces by choosing a certain option – i.e. the value of his or her next-best choice. The opportunity cost therefore measures the value of the alternative in a two-option choice. The text below gives a simple illustration of this.

The notion of opportunity cost.

A fan is planning to go to a football match. He puts the value of the event at €60, with a seat in the stadium setting him back €30. The same day, he learns that his favourite band is planning on giving a concert. He attaches a value of €80 to the attendance of this concert, with a ticket costing €40.

Let's begin by calculating the net values of these two options: going to the football match or the concert. In the first case, the net value of the cost is: €60 - €30 = €30, while the net value of the concert is: €80 - €40 = €40. The concert option, which provides a net value of €40, is therefore worth more than the football option which only provides a net value of €30.

What is the **opportunity cost** of going to the concert instead of the football match? It is the net value of the match – the value that the fan must renounce if he decides to go to the concert held on the same evening, so €30. The opportunity cost is what a decision-maker renounces when he chooses the best option from the viewpoint of its net value (his best choice), i.e. the net value of the value choice immediately below, or his next-best choice.

Although very easy to understand, the notion of opportunity cost is often the source of much confusion and error. The most common error is to liken the loss suffered by going to the concert rather than the football match – or the "opportunity cost" – to the difference between their respective net values, so €40 - €30 = €10. Although it seems fairly obviously that €10 is an extra benefit and not a cost, it is sometimes mistaken for the opportunity cost. This type of error is probably due to the fact that some authors speak of net opportunity benefit when referring to this €10, which is confusing. The "opportunity" in the case of the **opportunity benefit** refers to the existence of an option (the concert) which provides a higher net value than the football match, a sort of windfall effect, and not "the opportunity" to which the idea of opportunity cost alludes – this referring to the loss suffered because the most beneficial option was chosen (in this case going to the concert rather than the football match).

Figure 82

© J.C. Armand

The opportunity cost of going to a concert instead of a football match is the net value of the match – the value that the person must renounce if he decides to go to the concert.

There are two main purposes to assessing opportunity costs in environmental economic valuation:

- to assess the economic weight of restrictions introduced by an environmental policy on the stakeholders who must comply with it;
- to measure the value of an ecological service based on the cost of the artificial alternative that the same level of environmental quality would provide if this service didn't exist.

Let's look as these two scenarios in turn.

Assessing the costs created by an environmental restriction

In the first case, two situations of interest can be distinguished. In the first, the introduction of the environmental action will lead to restrictions for stakeholders, in the direct form of loss of economic benefit that can be calculated financially. In the second, the restrictions brought about by the environmental policy will incite changes in stakeholders' behaviour, which will have a cost for them in terms of what they choose to do before the environmental action comes into force. Let's give two examples of these two situations: planting grass along river banks and restricting river pumping in keeping with a policy for reserving flow for the benefit of aquatic ecosystems.

The example of grassy banks

For farmers, grassy banks (Figure 83) are similar to regulated fallowing. If the grassy banks are to be planted with the long-term in mind, which makes sense, the loss of agricultural benefits this policy creates for the land owner, per hectare of land that has thus become fallow, can be assessed based on either the price per hectare of farmland, or the loss of ground rent to which it gives rise. If the farmland market worked perfectly, the price of land and rent per hectare should be identical and it wouldn't matter which measurement was taken.

Figure 83

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Grassy bank policies are a good example of the use of opportunity cost methods. This cost is the loss of land value once agricultural practices have been reorganised following the planting of the grassy banks.

However, it is well known that the land market is subject to significant transaction costs, and thus to considerable differences between the transaction prices observed and rent levels. It would therefore be more appropriate to select the loss of ground rent as the closest measurement of the loss of agricultural benefit. This loss of ground rent is **the opportunity cost** of the fallowing of river banks in the sense of what the farmer renounces if he chooses to comply with the grassy bank policy.

If the surface areas concerned are modest compared to the overall surface area of the farm, the farmer will hardly change his cropping programme or farm management method after the grassy banks have been put in place. In this case, the annualised value of the gross margin per hectare – i.e. the gross margin amount per hectare divided by the interest rate of medium to long-term assets – expresses the opportunity cost per hectare of the policy for the farm owner.

If, however, the policy applies to a significant portion of cultivated land, the farmer will in all likelihood change his cropping programme or farming practices as a result. If this happens, **the measurement of annualised gross margin to take is the gross margin** after the farm's management procedure has been reviewed, and not the initial gross margin – contrary to common practice. This is because the opportunity cost measures what the farmer renounces in terms of his next-best choice. By reorganising his farm as a result, the farmer reduces the initial loss suffered by the grassy bank policy. The opportunity cost is the remainder to be deducted after this reorganisation or, in other words, the best possible gross margin through the best possible review of the farm's profitability with the grassy banks.

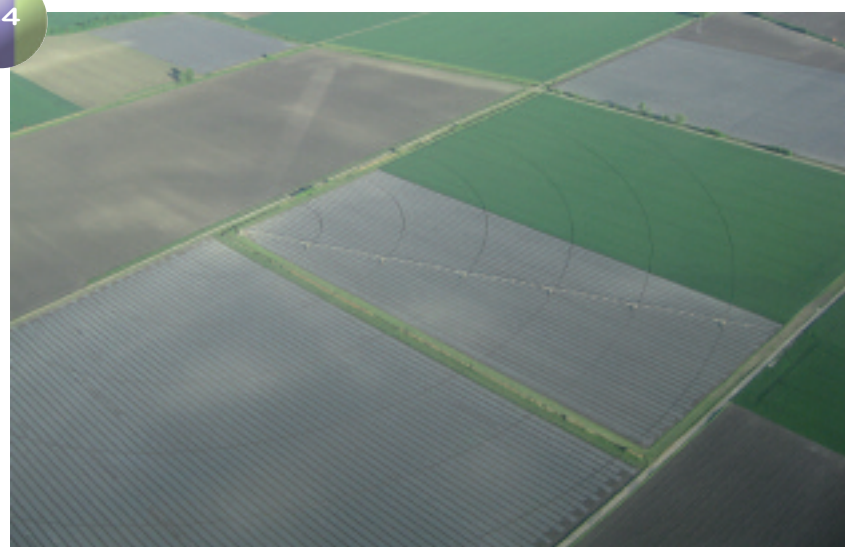
Of course, it isn't that easy to calculate such a gross margin after the planting of grassy banks because of insufficient data. Micro-economic models of agricultural production can help to get over this stumbling block by providing cost assessments once the optimum reorganisation of farms has been simulated. With no compensation, a comparison of the land transaction prices on the plots affected by the grassy bank policy with similar plots that are not subject to the policy can also give a close measurement of the opportunity cost.

The example of grassy banks can be applied to detention basins which concern the same loss of land benefit for owners – the main difference with the example above being that, in the case of floodplains, we should think in terms of loss probabilities, i.e. expected utility of the opportunity cost, also called risk premium. We will not describe the calculation method in this instance to avoid further weighing down a report in which we want purely to stick to the main points of the valuation method for ecological services.

The example of river pumping restrictions

In the case of river pumping restrictions for irrigation (Figure 84), the analyst faces the following problem. While land has a price – and therefore a market value – which serves as an indicator for calculating losses of benefit, or, in the same way, compares a ground rent equivalent to the annualised gross income of the farm, water does not because there is no water market. The opportunity cost of watering restrictions can therefore only be measured **indirectly,** by assessing the profitability variations caused by the changes in irrigation practices.

Figure 84



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Since water does not have a market value in its natural state, its value for agriculture can only be estimated indirectly

If the irrigation user consumes all of his authorised sampling quota, he may either renounce the right to irrigate more, or try and purchase more water – from a neighbour who hasn't used up all of his quota for example. The maximum price he is willing to pay for this purchase cannot exceed the loss of income resulting from the cap set by the quota, i.e. the opportunity cost of water access. In other words, the opportunity cost of a use restriction is **the maximum consent to pay to loosen the restriction** or, in a nutshell, the cost of the restriction. That said, remember that the opportunity cost must be determined assuming that the farm's irrigation and cropping programmes have been reoptimised following the pumping restrictions.

These two examples shed light on situations where the implementation of an environmental policy has restricted the practices of economic stakeholders. Note that these restrictions are expressed in quantitative form. The grassy bank policy sets aside a certain number of hectares along river banks, while a reserved flow policy reduces the flow pumped by irrigation users in the summer months. These quantitative restrictions can be associated with the subsequent monetary equivalents of the losses endured by consumers when the opportunity costs are measured. As they stand, these measurements are useful in negotiations on the grants or compensations to be allocated to owners to win their support for policies. At a more political level, they enter into the debate on choosing which measures to take. If environmental impact is the same, those measures creating the lowest opportunity costs for the community should be chosen by public managers.

For the purposes, this time, of cost-benefits rather than cost-effectiveness, the opportunity costs in the previous sense should be factored into the political choice and compared with the expected environmental benefits of the action. A collectively optimum policy should maximise the difference between benefits and costs, thus balancing out the collective marginal benefit and collective marginal cost. This is the line of thinking that sometimes leads to the environmental benefit being measured on the basis of the opportunity cost. However, take heed that such an approach implies collective efficiency. But if we consider an environmental policy, it's because we don't believe the current situation to be the best for society – otherwise why would we want to improve it? The initial situation in this case should not be considered optimum, as it is unsatisfactory in terms of environmental quality. As a result, there is no reason for a measurement of opportunity costs referring to a mediocre initial situation from the collective point of view to provide a measurement of the expected benefits from an environmental policy, since such a comparison is only possible once the ideal situation we want to achieve has been reached.

■ Measuring the value of a service by avoided costs

Having made these clarifications, let's now turn our attention to using the o**pportunity cost** method to measure the value of ecological services on the basis of avoided efforts for achieving an environmental objective. Annex 6 illustrates such a protocol being used to measure the value of the purification service by natural environments (Figure 85).

The annex describes the formal points of this process, but we'll look at a few key characteristics. The example given in the annex supposes that the public authorities are enforcing a river quality standard. Achieving this objective supposes expensive sanitation efforts. It is thus possible to calculate an opportunity cost of such a restriction on the basis of the cost of these efforts. By protecting the purification ability of environments, we reduce this opportunity cost. The variation in opportunity cost resulting from an improvement of this service provides an economic valuation of the service. The example does show, however, that this measurement depends to a large extent on the type of purification process at work in environments.

Figure 85

a © B. Suard – MEDDTL
b © L. Mignaux – MEDDTL

By protecting the purification ability of environments, we reduce the opportunity cost of a river water quality objective.

Given the type of problems associated with the ecological management of water and associated environments, we can immediately see **the sheer potential for application of this notion,** both for the development of rivers and the management of environments in general. Complex interactions are at work within hydrosystems. For specific environments or specific segments of watercourses, assessing the opportunity costs identifies the benefit to be expected from management measures applied not just in the environment in question but also other environments interacting with it. By construction, this benefit also takes account of the impacts of the measure in question on all the uses concerned – as well as users' reactions to the measure.

■ Main limits to the method

What are the main limits to this method? First of all, **the measurement greatly depends on the restrictions taken into account.** Typically, neglecting to consider restrictions or inter-dependencies can lead to significant errors of measurement. This is shown by opportunity cost approaches based on integrated models of water management when they are compared to measurements of impacts for specific users. Secondly, the concept's relevance is derived from the idea that public managers and private stakeholders alike play an optimising role, managing the restrictions to the resource at their disposal as effectively as possible. But these managers and stakeholders know only so much about the system and its interdependencies and must also deal with random and uncertain phenomena.

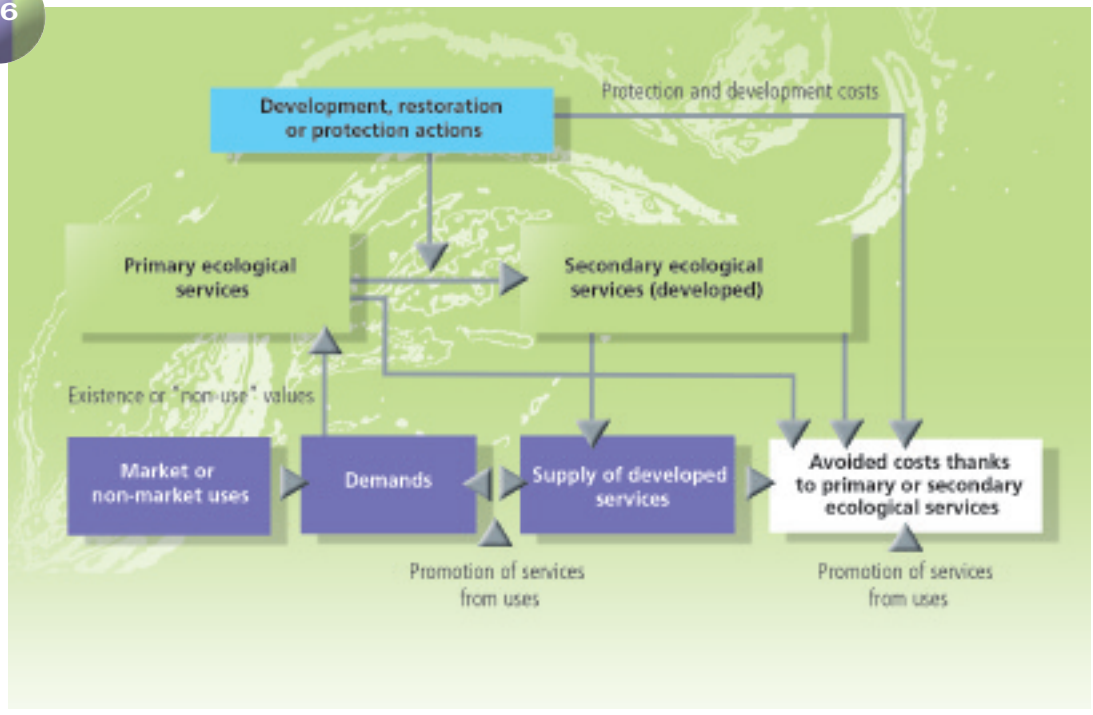
This is how studies on the management of dams in California concluded on ineffective management by the operators, as the decisions that were actually made did not conform to the economic optimisation models – hence the accusations of "irrationality" and "inefficiency" levelled against the resource operators. Subsequent studies going into greater detail and factoring in the specific limits **to managers' decisions** due to uncertainty have righted this diagnosis and shown that, considering such limits, the managers' apparently "irrational" decisions were in fact perfectly coherent and understandable in the context they found themselves in.

In other words, the limits specifically affecting the stakeholders' decisions do not allow for an optimisation balancing out the opportunity costs of these limits and the subsequent value of ecological services. A poor specification of the actual nature of the management problem facing decision-makers will result in them making ineffective choices and advocating impracticable or unsuitable measures. This poor specification will also bear on the measurement of the associated opportunity costs and therefore the valuation of ecological services. As always in the valuation field, the resulting figures give us an idea of orders of magnitude and priorities, but we shouldn't lose sight of their fragility or strong dependence on hypotheses when establishing them.

Summary

To sum up (Figure 86), economic valuation methods can apply to the market or non-market **demands** and the **supply** of ecological services, by taking an opportunity cost approach in this case. In all cases, they produce **monetary valuations** of the supply of different services for society. As mentioned above, these "services" – connected for the most part to uses – are, in fact, secondary (or "developed") services stemming from the ecological services as defined by the MA typology. At the micro-economic level, it will on the whole be impossible to work back from the value **of secondary services** to a monetary attribution for each primary service, since these primary services form a whole from the point of view of socio-economic promotion.

Figure 86



Economic valuation methods can apply to the market or non-market demands and the supply of ecological services, by taking an opportunity cost approach in this case.

The merits of **tackling economic valuation from the angle of the entire hydrosystem really stand out** in this instance. Protecting a given local environment within a hydrosystem, just as in a complex of environments in the context of ecological solidarity, alters the supply conditions for all of the primary ecological services produced by these environments. When assessing the ecological impact of measures being considered, we thus already find ourselves in **a mindset of primary service composition.** We can associate another complex of secondary services to this specific complex of a hydrosystem. It is the worth of this complex of secondary services to society that will be estimated. A matrix of uses, and thus of users, can be associated with this complex, and economic valuation methods can deduce the value of these uses by analysing the users' behaviour.

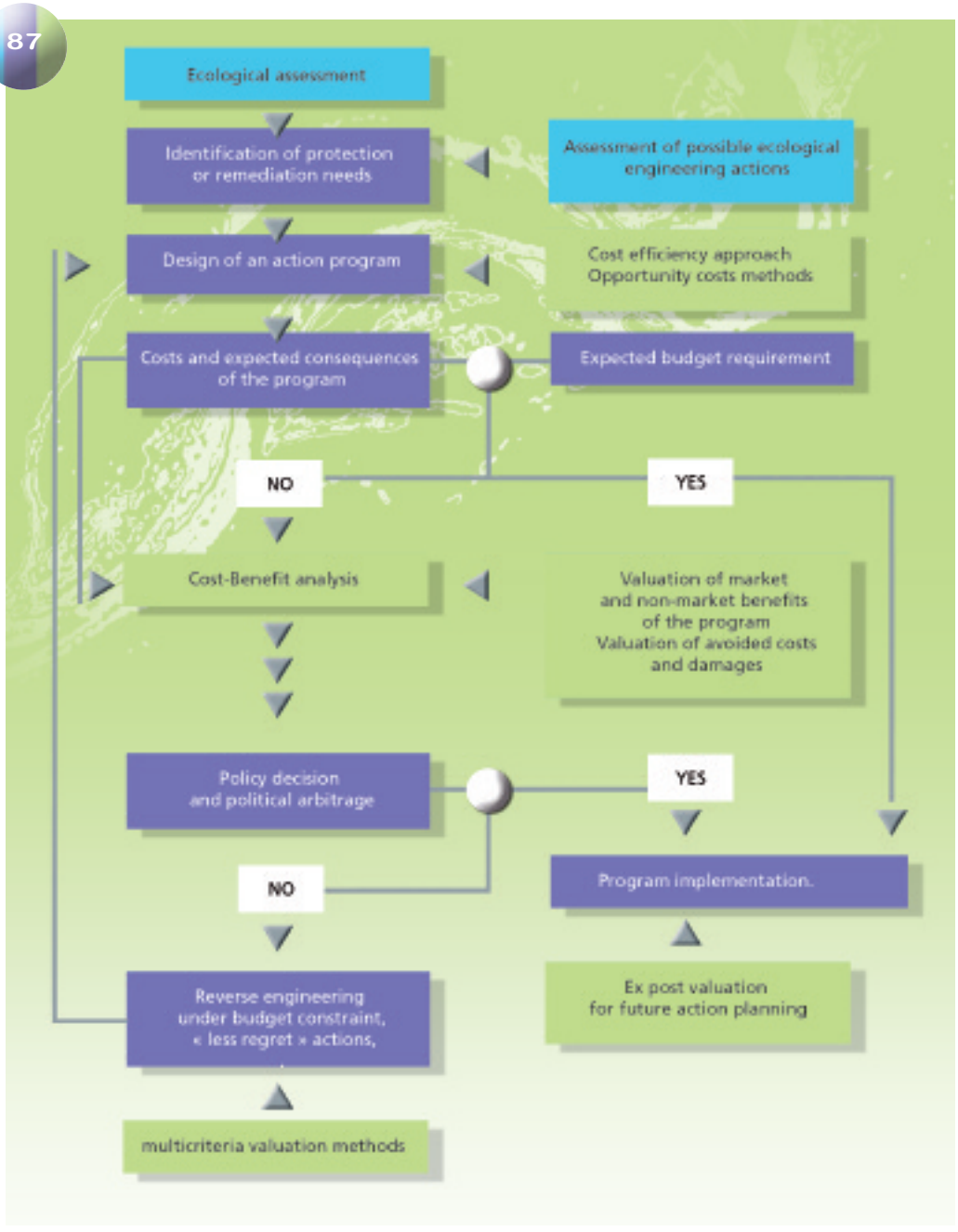
As we have already noted, the approach is variational: the point is not to measure the "actual" value of ecological services. Such a measure would boil down to the comparison of one situation – in which natural environments exist – with another, in which they don't exist, which is absurd. But in a measurement of the benefits associated with the measures aimed at achieving good status, the **comparison weighs the current status against the expected status.** The benefit is therefore, in fact, what the society gains in monetary equivalents from the changeover from one status to another. The point is consequently to measure value variations associated with this improvement.

We can perfectly well reverse this focus too and, instead of talking about the benefits associated with the improved status, look at the cost incurred by society because of the current poor status of environments – which we call "environmental" damage. Transposed into the field of economic valuation methods, we can see that the changes in behaviour brought about by the improvement in status are the focus of the assessment: changes in visitor numbers or land values near the protected environments for example.

Conclusion

We have touched on certain operational implications of the valuation of services on several occasions. It seems fitting at this stage to attempt an initial summary of these. So let's imagine we have a hydrosystem to manage. The multiple-stage approach suggested in Part 2 is meant to have been implemented and divided up the coherent space into ecological and management units. Let's call these spatial units "ecological management units".
An operational valuation protocol could be organised around the sequence described in Figure 85.

Figure 87



Operational protocol for assessing the value of services in the context of a hydrosystem to manage.

1. Ecological diagnosis

Using the quality criteria, we can make an initial diagnosis of the ecological status of the hydrosystem and distinguish which units need improvement from those whose status is already good. Diverse complementary studies have explained the causes of poor status and the determining factors of good status of different units. As we have already highlighted, the approach taken to the large-scale water cycle is not just curative. Prevention is also necessary to avoid deterioration of healthy environments.

2. Identification of protection and restoration methods

Based on the previous diagnosis, this stage seeks to identify the most appropriate measures for restoring good status or protecting it as necessary. It therefore calls on ecological engineering and must carry out an initial calculation of the estimated cost of different measures.

3. Design of a programme of measures

At this stage, we need to decide between different measures for each ecological management unit with the aim of drawing up a programme of measures for the whole hydrosystem. The WFD recommends using a cost-effectiveness approach to construct this programme of measures. We give a formal description of the steps involved in implementing the cost-effectiveness approach in the annex. This must ultimately identify the programme(s) of measures most likely to achieve good status for all the management units at the least cost overall. It therefore produces an economic calculation of the total cost of the most cost-effective programme of measures.

4. Cost-benefit approach

It the budget earmarked for protecting the hydrosystem exceeds this cost, all we need to do is implement the programme of measures previously identified. But this may not be the case. If not, we need to look at the socio-economic benefits of the cost-effective programme of measures identified at stage 3 (Design of a programme of measures). This is when a valuation of the ecological services should be performed.
This will put a financial figure on the ecological benefit of the action being considered.
Two points should be made at this stage:

- the valuation does not bear on the ecological services provided by the hydrosystem "in general", but only on the benefits in terms of improvement of the services provided that are associated to the cost-effective programme. This is in fact the cheapest way of achieving good status. It is thus unnecessary to look at possible alternative programmes which, although more expensive, would produce more benefits;
- the valuation exercise focuses on a programme of measures rather than the actual ecosystems. For all that, the temptation to ask individuals their opinion on the measures within the programme, through a contingent survey for example, should be resisted, as the general public does not have the necessary skills to make such judgements.

A study conducted for EDF on the burial of electricity lines is an "enlightening" example of this. Taking the contingent approach, the study asked users about their willingness to pay to bury X kilometres of lines per year by providing the cost for EDF of one kilometre of lines buried. The respondents found it extremely difficult to picture the difference between 1,500 km of buried lines per year and 10,000 km.

Many of them were even shocked by the high burial cost and called for the programme to be dropped to reduce their electricity bill – a reaction which the questionnaire designers had not foreseen in the slightest...

On the whole, participants in contingent surveys have no concept of the financial orders of magnitude of a public policy, nor of its technical scope. While "lines" (electricity lines or rivers) naturally mean something to engineers, they don't ring any bells for the general public. The study should only present the expected impacts of the action being considered, and the valuation bears on these impacts.

5. Political arbitration

Armed now with a figure for the least expensive way of achieving good status and for the associated ecological benefits for society, the managers can now enter into a more political stage as to whether or not the programme of measures can be financed. It is at this stage that the calculated value of the benefits should be communicated to the decision-makers to help them come to a final decision. A discussion on the notion of "disproportionate" costs should also be held at this point. Two possibilities arise. Either the decision-makers choose to finance the programme and consequently increase the budget ring-fenced for the hydrosystem, or they refuse. In the latter case, a reverse engineering approach might be taken, aimed at identifying which programme of measures minimises the ecological management units' deviation from the good status – for a given budget. The difficulty of this approach is that there is no reason to consider the 80% rate of good status achievement for a specific management unit to be equivalent to the same percentage for another management unit. This leaves us with a complex choice of weighing up the tolerable deviations from the good status for the different environments making up the hydrosystem – a classic challenge for multi-criteria operational research.

Lastly, the programme must be implemented alongside an ex-post valuation of its consequences – a crucial step with regard to future plans. This valuation will need to be budgeted upstream, from the programme outset.

The succinct presentation of an operational valuation approach in key stages sheds light on the fact that there's no reason for service valuation to be systematic or the main criterion for decision-making. The political decision-making process described in stage 5 (Political arbitration) will inevitably touch on countless budgetary considerations besides the sole protection of a specific hydrosystem's natural environments.

For environmental valuation will only prove completely worthwhile for political decision-making purposes in the more general context of the development of practices for assessing public policies.