A SAVOIR

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Tools for what trade?

Analysing the Utilisation of Economic Instruments and Valuations in Biodiversity Management

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À Savoir

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Contents

Introduction

Over the years there has been a dramatic surge in the attention given to economic tools for biodiversity (ETB). Promoters from academia, government or NGOs are advocating them as the solution that may turn around the presently losing battle against biodiversity erosion. They underline their expected strengths: their flexibility (compared with regulatory tools), their ability to put biodiversity on a par with other political priorities by internalising it into economic reasoning and their potential to increase massively the level of funding available for preserving biodiversity. They are, however, contradicted by numerous and vocal critics, who are accusing economic tools of commodifying nature, of reinforcing the very economic system that threatens biodiversity, of imposing artificially economic notions on environmental issues.

Focusing on the actual use, rather than the principles, of economic tools for biodiversity

As one reads the literature on ETBs, or as one is involved in the daily discussions they trigger, it soon appears that the attention is focused on the principles that underlie ETBs. Much less attention is paid to their actual use, and much less evidence is available, despite brave efforts to identify pilot projects and success (or horror) stories.

A crucial issue for practitioners, for critiques and for supporters of ETBs

We now believe that observing and understanding the actual uses of ETBs should become the priority of the field, whether one holds a supporting or a critical position of economic approaches to biodiversity. From the point of view of those who advocate ETBs, being able to demonstrate that they are (or really can be) actually and effectively used on a large scale is a necessary condition for the credibility and implementation of their argument and plans of action. But investigating actual use of ETBs is just as important to those who criticise them. The grounds are more varied here, because they depend on the type of critique they relate to. For instance, if one fears that implementation of a given tool may have ethically or politically negative consequences, if the tool is not actually used (or usable), such fears are in effect pointless... Moreover, whatever the extent to which they are used, the possible negative consequences of ETBs depend to a large extent on the context and the way they are implemented.

As for those critiques that focus on the general principles underlying ETBs – denouncing for instance commodification of nature – connecting such *a priori* critiques with actually observed empirical effects should be of interest, and this has to rely on the study of actual ETB use and its consequences.

Understanding the actual use of ETBs is important also for those stakeholders and operators of the biodiversity field who are neither advocates nor critics of ETBs but observers or declared pragmatists seeking positive results for biodiversity from the entire panel of available tools, economic or not.

For all these reasons, it is high time to become serious about understanding actual use and usability of ETBs: how they are used, by whom, to what extent, with what results. The use of ETBs being such a central issue, one would expect that it would already be the object of much research. However, the literature on this topic is quite limited both quantitatively and in terms of the scope it covers.

Use and implementation of valuation: a major frontier

As far as economic valuation is concerned, there is little documentation in the literature about its use in actual decision-making (Laurans *et al.*, 2013). What evidence there is suggests that it is rarely used by decision-makers, despite the fact that such evaluations have been available and their usefulness advocated for over thirty years. Along with other authors who have recently reviewed the field (see for instance Liu *et al.*, 2010), we feel that dealing more clearly with issues of *use and implementation* is the major frontier for the further development of biodiversity valuation.

Economic instruments for biodiversity: a gap from theory to actual use

In the case of economic instruments that actually put money on the table, the issue is different. There is already a significant amount of practice and experience with such instruments as payments for ecosystem services, conservation easements, biodiversity banking, etc. Here, the gap lies rather between the theory providing the rationale and design for such instruments, and the current dynamics and contexts of their use (Vatn, 2009). Vatn underlines, for example, that there is a major contrast between the market-based argument underlying much of the advocacy in favour of payments for ecosystem services and the fact that observation of most cases of implementation show them to be intricately embedded in the administrative management of environment. One may shrug off such findings with a joke often pointed at those who are inclined to reasoning too much in the abstract: "I know it works in practice, but does it work in theory?" The point is that what works in practice in some contexts, and/or on a small scale may very well not work in other contexts or on larger scales, if one does not really understand why it does or doesn't work.

This is precisely the challenge now facing the development of economic instruments for biodiversity. If they are to contribute significantly to dealing with the biodiversity crisis, there are important questions about their use that require answers. How well and under what conditions can they transfer from one place or one issue to another? To what extent and how can some of them become generalised, and can they work usefully on scales massively larger than now? What are their negative impacts and the problems they create in practice? Such questions require not only being able to post up a limited number of pilot ETB programs, but also acquiring an in-depth understanding of the actual dynamics at play in the use of economic instruments in real-world biodiversity management and policy contexts.

The first aim of this book is to make the case for a greater awareness and much more research on the actual use of economic valuations and instruments for biodiversity in real management and policy-making contexts.

Mobilising appropriate theory to account for management and policy contexts of ETB use

The value of this, however, would be limited if new ways of addressing the issues of ETB use were not put forward.

The complexities and ambiguities of social and political contexts are an inherent, not an avoidable, characteristic of real world ETB use situations

When reading publications on ETBs, again and again, we have come across statements to the effect that the clever design of economic tools and the positive outcomes we could expect from them, were they implemented without interference, are hindered by social and political processes, both irrational and objectionable. ETBs, however, are always necessarily, not accidentally, used within decision situations and action systems that are essentially social and political.

From our point of view, the still limited purchase of the literature dealing with issues on ETB use is not due solely to a lack of awareness or attention. It is also linked to the fact that the theoretical frameworks, concepts and models on which most of the literature on ETBs is based lie mostly in economics. Economics provide interesting resources in the understanding of political, social, organisational and cultural dimensions of biodiversity issues, but these resources are limited in scope. One way forward is for economists to gradually open up their perspective to include social and political aspects (an opening up that is at the basis of the "ecological economics" movement). Even if internal academic struggles within economics currently tend to slow down this interdisciplinary opening up and to confine it to the academic margins, this line of work has proven to be quite useful. But its potential for further development is also limited by the intrinsic limits of economic theories in grasping the complex management and policy contexts. As we shall see in further parts of the book (especially, chapter 7), the quest for continuity with the paradigmatic foundations of economics comes with limits on the scope of what political and managerial dynamics can then be taken into account.

The need for specific frameworks and theories

In line with these observations, to understand the challenges involved in the real-life use and biodiversity outcomes of ETBs we have to develop a larger repertoire of conceptual resources to use and grasp more widely and more precisely the dynamics of the social, managerial, and political processes of actual use of economic instruments for biodiversity.

If the internal rationales of ETBs are essentially grounded in economic theory, we believe that the rationales of their uses also have to be grounded, and just as deeply, in managerial, social and political realities and theory. We propose here to pause and look afresh at what theoretical resources can be used and how they can provide a sound basis for examining the places and roles of ETBs in the actual processes of biodiversity management and policy.

Making use of the plurality of theoretical perspectives

As we embark on this exercise, it is immediately apparent that there are many different theoretical perspectives offering accounts of the processes of policy and management, and thus of biodiversity policy and management. Those perspectives are very different from one another, and sometimes contradictory. This plurality of perspectives is inevitable when discussing social and political processes. But although it may make matters more difficult, especially for those who would prefer to discuss only within the frame of one unitary theory, this plurality is actually very resourceful. Allison and Zelikow were among the first to make this point quite strikingly in the 1964 book where they present three very different theoretical interpretations of the US government decision-making process in the Cuban missiles crisis (Allison and Zelikow, 1999). In this pioneer work, they show (1) that each of the three theories they use makes a unique contribution and that they complete one another, each clarifying specific aspects of the decision-making processes; and (2) how their confrontation provides an additional contribution to our ability to account for complex political decision-making processes.

Since then, we have repeatedly experienced the fruitfulness of deliberately cultivating this plurality of approaches of environmental – and especially, biodiversity – issues. So as we undertake here to engage in a dialogue with the internal, economic rationale of ETBs, we shall not choose one perspective and develop it at length. We will rather examine several perspectives which we have repeatedly experienced in our respective research and professional activities as having much potential for analysing and designing strategy in complex environmental management situations. We shall examine each perspective thoroughly in order to extract some useful insights for practice and to probe its potential for further, in-depth research on ETB use issues.

Stepping out of a priori

Indeed, stepping outside of economic theory and looking at ETBs from a social or political perspective is not new *per se*. It is the position adopted also by the more radical critics of the use of economic tools for dealing with environmental problems. In his book on *Markets, Deliberation and Environment,* John O'Neill (2007), for instance, systematically presents fundamental antagonisms between the social and political logics of environmental problems and those of the market and of the economic tools grounded in it.

Our argument here is that such stepping out and looking at economic tools from social, managerial and political perspectives is just as useful for promoters of ETBs (for a closer-to-the-ground understanding of their use) and for biodiversity operators and stakeholders in general (to get a grasp of ETBs in the context of actual biodiversity management situations) as it is for critics of economic tools. Research on uses of ETBs will not be able to develop fast enough, or far enough, if it remains predicated on postulating either an *a priori* compatibility of economic and social-political logics, or an *a priori* incompatibility between them. The roots of our own work lie in the study of complex environmental management systems (Mermet, 1992; Gaudefroy de Mombynes, 2007) and decision-making processes (Laurans, 2000; Laurans and Cattan, 2000; Laurans and Dubien, 2000; Mermet, 2003; Mermet, 2005). Such management systems and processes combine very different dimensions and rationales, and hybridise extremely heterogeneous elements, mechanisms and perspectives. So we will advocate here that attention and research should be focused on how, in the uses of ETBs, these hybridations function and why they fail or work. This relies on observing *in concreto* how that use (or non-use) works. But it also requires that we identify and use appropriate theoretical resources for the description, analysis, evaluation or design of these hybrid systems themselves. The most important part of our work on ETBs has been devoted to this theoretical effort.

To sum up, the aim of this book is (1) to identify the issues raised by the use of ETBs, (2) to advocate that research on such use should become a priority on the biodiversity research agenda, and (3) to show how actively mobilising a plurality of theories on the mechanisms and designs underlying biodiversity management and policy-making could lead to breakthroughs that are much needed for the future of the field.

An extensive scope both on biodiversity and on economic tools

Before indicating by what methods we have pursued these goals and presenting the outline of the book, an explanation may be necessary on the scope covered here.

Biodiversity has become an extensive concept, covering most environmental issues connected to ecosystems

In terms of the environmental issues covered, we will address all the facets of biodiversity. Although it has been used only for a couple of decades, the concept has come to encompass most environmental issues connected with ecosystems and ecological landscapes, as well as their structure and way of functioning. It includes the genetic diversity of organisms and their various assemblages, but also the "ecosystem services" that accrue to societies from the functioning of ecosystems – for instance in terms of water purification or flood mitigation, but also ecosystem-generated resources, such as fisheries or forest resources. We are well aware of the controversies between different perspectives that adopt a more or less narrow scope, or that concentrate either on ecosystem services or on non-utilitarian values of biodiversity. Such choices of scope, however, are part of the strategic choices that are made in the use of tools and in the analysis of biodiversity management situations (see for instance the various scales of values analysed in chapter 8). To be able to examine the various options in our analysis, we have adopted the widest possible scope, including all the facets of biodiversity, from ecosystem-based resources to the conservation of genetic diversity at all scales.

We think it is also important to keep in mind that behind the apparent novelty of notions like "biodiversity" or "ecosystem services" lie concepts, positions, practical issues and strategic perspectives that have been present for much longer. The importance of conserving "life supporting" ecosystems was already the central theme of the 1980 IUCN World Conservation Strategy [1], for instance. And in his very sobering writings, Patrick Blandin (2009) shows strikingly how all along the 20th century conservation organisations have been divided between those of their members who were in favour of focusing public attention on the usefulness of nature for human interests (today's "ecosystem services") and those who feared that such a focus would put those components of biodiversity that had no immediate human utility at risk.

To us, both sides of this debate have obvious merits. We shall not restrict the scope *a priori,* and we will mostly use the various notions (biodiversity, ecosystem services, etc.) as they are habitually used currently in the mainstream literature on ETBs, and introduce more precision on definitions only in places where they are really necessary.

Discussing valuation and "cash on the table" economic instruments jointly

In terms of economic tools our scope will include both valuation and "cash on the table" instruments. Ecosystem Services Valuation (ESV) has been the object of much (research and) experimentation over the last 30 years and has generated an abundant literature and body of expertise. As for economic policy and management instruments – those that use money to induce changes in agents' behaviour towards biodiversity – there is a variety of them (to name only a few: payment for ecosystem services, offsets, quotas, conservation concessions, etc.), and they have been the object of numerous experiments over the last decades.

Most of the time, the two sets of tools (and often, the various policy instruments) are discussed separately and treated as separate fields of action, expertise and research. Here, we will consider jointly the whole range of (ETBs). Of course, we do see how their fields of practice and research differ in some important respects. But they are connected, on the one hand, by deep theoretical links (in particular when grounded in economic theory), and on the other hand, by issues in use and implementation. To provide just a few examples, the case for valuation methods rests largely on the potential use of valuation results for implementing economic policy instruments

[1] http://data.iucn.org/dbtw-wpd/edocs/WCS-004.pdf

such as taxes, payments for ecosystem services, or offsets (Liu *et al.,* 2010 2010). Discussion on the limits in implementing policy instruments, in particular payments for ecosystem services, often focuses on the difficulty of securing reliable economic valuations (see for instance Karsenty *et al.,* 2010; Landell-Mills and Porras, 2002). In other words, valuation and instruments relate to the same understanding of the biodiversity policy-making issue, and to the same judgement about the solutions to promote. They belong to the analytical and operational aspects, respectively, of the same action framework: they consider implicitly that the problem is that of a correction of market failures by providing effective economic signals (in the form of information in the case of valuation, of actual money in the case of instruments). Since the project is to focus on the use and implementation of ETBs, and since it will also re-examine the links between practice (that is, use and implementation) and theory, we feel the connections between the two subfields are essential, and it is justified here to treat them jointly.

A multi-faceted approach

The research that led to this book started from material and insights already available through previous work involving the authors. Of particular notice are:

- a project on valuation of biodiversity, led by IDDRI and funded by the Hermès foundation, that focused on the use (or indeed, non-use) of such valuations, through literature reviews, seminars and case studies,
- a project on payments for ecosystem services led by Yann Laurans and Tiphaine Leménager and funded by AFD, reviewing the great diversity of PES mechanisms and their current development, (Laurans *et al.,* 2012)
- several projects led by Yann Laurans on valuation of ecosystem services rendered by French wetlands (Laurans *et al.,* 1996; Laurans et Dubien, 1996; Laurans, 2000; Laurans et Cattan, 2000, Laurans, 2009, Laurans et Aoubid, 2010)
- former studies led in the 1980s and 1990s by Laurent Mermet, on valuation of biodiversity and on the use of cost-advantage studies in decision-making when the environment is at stake, providing some temporal depth to a field that seems ever ready to present itself as new.

Expanding from the material and insights already accumulated, the research itself consisted of four complementary approaches.

1. The first was a critical review of the literature. The central aim of this component was to re-examine the literature on ETBs with a deliberate focus on utilisation and implementation issues, and on the theoretical resources which are – or are not – mobilized to analyse them and to explore difficulties and solutions.

2. The second was a series of interviews with persons involved in the use of ETBs. We focused on three groups: economists in organisations involved in conservation or in development, project operators of conservation projects, plan or programs, and policy decision makers. We discussed with the interviewees about their interest in ETBs and their own experience with ETB implementation through detailed stories. We also tried to understand if they were considering ETBs as a new solution, if their organisation had taken a specific stance on it and if they were facing specific debates concerning ETBs.

3. The third component of the project was the systematic consideration and discussion of various perspectives, frameworks, conceptual models, and theories that underpin the analysis of environmental decision-making and management processes. Here, we drew on our wider research on theoretical resources for the analysis of strategic issues in environmental management [2].

4. This fed into the fourth component of the project that consisted in regular in-depth discussions within the research team and with other experts through the organisation of a seminar series entitled "The Economics of Biodiversity: Practical Solutions or Esoteric Diversions?", including presentations by senior researchers on ETB use issues [3].

Book outline

There are ten chapters in the book. The first four, based on the literature reviews and interviews, give an overview of utilisation issues raised by four types of economic tools for biodiversity. Chapter five draws from that material a set of organising questions to guide further work to better understand issues and contexts of use of ETBs. The last five chapters examine these questions systematically and propose a set of theoretical perspectives and resources that we think have great potential both for giving a better practical grasp of ETB use situations, and for further research.

[2] See for instance: laurent-mermet.fr

[3] http://www.smithschool.ox.ac.uk/the-economics-of-biodiversity-practical-solutions-or-esoteric-diversions/

A more detailed presentation of the book's outline.

As regards the first four chapters, we have adopted a grouping of ETBs into four main types, according to the basic kinds of economic and managerial operations they accomplish for the operator using them to manage biodiversity: valuation of ecosystem services (ESV), payment for ecosystem services (PES), buying land or land-based rights and offsets and biodiversity banking.

Chapter 1 discusses valuation of ecosystem services (ESV), that is, tools that attribute a monetary value to biodiversity, aiming at taking this into account in the decision-making process.

Chapter 2 considers payments for ecosystem services (PES), *i.e.* economic instruments that consist in paying a rent for biodiversity. These are instruments where actors, aspiring to better biodiversity, pay for conservation, management or restoration of an ecosystem on a continuous basis.

Chapter 3 is devoted to those instruments where operators directly buy land or land-based rights for biodiversity conservation: through a one-off payment, they acquire long-lasting rights on land that transform the economic situation and guarantee some form of conservation. This includes the purchase of land for conservation, conservation easements, and conservation concessions.

Chapter 4 examines instruments that organise trading in biodiversity conservation and restoration, for instance through offsets, bio-diversity banking or biodiversity credits, and which stem from a polluter-payer principle.

These four types of tools are at very different stages in their development and raise quite different utilisation issues, so we have organised each chapter around what we consider to be the main, organising utilisation-related problem in each particular field.

Based on the systematic overview of the "toolbox" in the first four chapters, chapter 5 discusses the most decisive issues in the use of ETBs across their different types. These issues revolve essentially around how ETBs are embedded in legal, social, political, managerial systems of biodiversity management. We discuss the way the state of the art in the field, as reflected especially by The Economics of Ecosystems and Biodiversity (TEEB) report, views the wider context of ETB use. Building from there, we propose a set of five organising questions to improve our understanding of the situations and stakes of ETB use.

In each of the chapters that follow we examine one of these questions more closely, propose and discuss a specific theoretical perspective to analyse it in-depth, and derive useful diagnostic frameworks.

Chapter 6, addresses the question of agency in biodiversity conservation and in the use of ETBs. The relevant questions here are: "Who is in charge of collective action for biodiversity?" or "Who are the users of collective action tools like ETBs?" In our view, all positions on environmental management and decision-making, be they lay or scholarly, are based on a very limited set of possible concepts of agency in collective action – which we call action paradigms: government, coordination, revolution, governance and minority intervention. We will introduce these paradigms and discuss how each of them implies very different expectations from, opportunities for and issues in, the use of ETBs.

Chapter 7 asks to what extent agreeing on new rules and institutions is the way to better biodiversity management. It mobilises common-pool resources (CPR) theory, as introduced by Elinor Ostrom (1990). We show the deep influence the theory exercises on current thinking within the ETB literature and the deeper theoretical roots it provides for extending further some of the currently influential trends of thought about ETBs – especially the focus on institutions and participation. But we also examine the limits that are inherent to the CPR framing, and thus to the approaches, in the field of ETB use, that rely on the same foundations.

In chapter 8, we turn to the questions and controversies about values that are such an important part of ETB debates and field use. To shed light on them, we use Boltanski and Thévenot's justification theory (1991). This is an approach that squarely removes the quest of self-interest from centre stage to focus on the critique and justification of decisions on normative grounds. As we shall see, one of its valuable contributions for analysing ETB use is its capacity to grasp the normative dimension of the market while firmly positing alongside with other, incommensurable orders of worth.

Chapter 9 addresses questions of innovation and of the associated political dimension. To approach them, we present two theoretical models pertaining to the wider set of actor-network theory: Michel Callon's sociology of translation (Callon, 1986) and the political philosophy proposed by Bruno Latour in Politics of Nature (2004). These models have their own way of accounting for the way economics are woven with science, technology, politics, etc. into the fabric of social-ecological systems that are constantly being re-negotiated.

In chapter 10, we focus on the issues of strategy and power involved in trying to make some actors change what they do, for the sake of biodiversity. For this we use our own framework for strategic environmental management analysis (Mermet, 2011). This time, attention is drawn away from collaborative approaches and turned to a serious examination of the asymmetrical and often adversarial dimension of environmental issues. In many cases, conserving or restoring biodiversity depends on putting up a struggle to make some actors change their behaviour. And this requires strategy in the strong sense of the word, *i.e.* involving not only making action plans, but also facing opposition that intelligently deploys resources to try and make plans in favour of biodiversity fail.

Finally, an extensive general conclusion will synthesise the main findings and discuss the perspectives for further work.

Readers with different backgrounds may adopt different reading strategies. Those who are familiar with ETBs in their various forms can jump directly to chapter 5, start with the questions we think remain on the table considering the state of the art of the ETB field, and continue with the deeper treatment we propose in the following chapters. After that, they might return to the chapters on tools with a keener understanding of why we point to somewhat different aspects of them than the literature usually does. Readers interested in biodiversity management and policy but unfamiliar with economic tools (or familiar with only some type of tools) may want to follow the overall outline of the book and start with the first overview chapters, reading first on the principles and practical issues of the tools and then going on to the deeper reflection on biodiversity management and its use of tools in chapters five to ten.

1. Ecosystem services valuation: understanding and overcoming the implementation gap

Valuing ecosystem services to factor them into decision-making has been one of the proposals on the conservation agenda for well over fifty years (Gosselink *et al.*, 1974). In its most straightforward form, the reasoning behind the proposal is that (1) decisions that affect biodiversity are based on balancing economic costs and benefits attached to various alternatives; therefore (2) not assigning monetary values to ecological issues is tantamount to conferring them a nil value, and thus barring them from being taken into account in serious decision-making; thus (3) ascertaining and communicating the economic value of ecosystems in monetary terms is essential for them to be given due consideration in decisions and policies.

Putting this reasoning into practice, economists have proposed various methods, which have become the object of an abundant literature for valuing ecosystem services in monetary terms. Indeed, the main methodological options have been on the table since the early 1960's (Gomez-Baggethun *et al.*, 2010). Up to the present time, this literature has kept on expanding, both in terms of methodological refinement and of multiple case studies applying methodologies to various kinds of ecosystems in different contexts.

In the last decade, valuation of ecosystem services has also become a more and more visible part of the conservation agenda. Its place in the MEA (Hassan and Scholes, 2005), the subsequent TEEB report (Sukdhev *et al.*, 2010; Wittmer *et al.* 2010) and the latter's consideration at Nagoya (see Decision IV/10 of COP 10 [4]) are familiar indications of this at the global level. Similar efforts have also been made at national or subnational levels, as illustrated by national experts reports (Chevassus-au-Louis *et al.*, 2009; UK National Ecosystem Assessment, 2011) or by the numerous environ-

[4] http://www.cbd.int/doc/quarterly/qr-10-en.pdf, checked May 2011

mental institutions and organisations which have recently increased their effort to account for the economic dimension of their activities (IUCN *et al.,* 2004; Parcs Nationaux de France, 2008; EFTEC and DEFRA, 2010; World Resources Institute, Hanson *et al.,* 2011).

High hopes were actively expressed a few decades ago that this collective effort for valuation of ecosystem services could be decisive in facing the biodiversity crisis, as when Pearce and Moran mentioned, in 1994, that the need for environmental economists is to "*be instrumental in altering decisions about [the use of natural assets], particularly in investment and land-use decisions which present a clear choice between destruction or conservation*" (Pearce and Moran 1994). Those are enduring hopes, as testify Daily *et al.*: "*Over the past decade efforts to value and protect ecosystem services have been promoted by many as the last, best hope for making conservation mainstream – attractive and common place worldwide. In theory, if we can help individuals and institutions to recognize the value of nature, then this should greatly increase investments in conservation, while at the same time fostering human well-being*" (Daily, Polasky *et al.,* 2009).

It is far from evident, however, that we are really about to see this breakthrough become a reality, as expressed recently in Liu *et al.*'s ex-post review of the use of a series of ESVs (2010): "*along with other reviewers, it was found that the contribution of ESV to ecosystem management has not been as large as hoped nor as clear as imagined*".

Whereas ESV methodologies have considerably developed over the last four or five decades, how they are actually used, for what results, and what influence, is still a question, and this question has not been addressed extensively until recently (Laurans *et al.,* 2013). In this chapter, we will probe the implementation gap that currently exists between the claimed usefulness of ESV tools and them making a tangible difference in solving environmental problems. We will first recall briefly the main tools in the ESV toolbox, and the main general critiques that have been addressed to ESV. We will then see that the debate between ESV promoters and critiques is now moving beyond the rather abrupt confrontation of unqualified support of ESV and no-holds attacks against them. The next two sections of the chapter will then look successively at the expectations raised by ESV and at the surprisingly scant evidence of use. Drawing lessons from the past and examining the options that are now on the table to improve the use of ESV, we will then turn to the challenge ahead: understanding and bridging the gap between the rapid development of ESV on the supply side and its actual use in solving biodiversity problems.

1.1. ESV: a well-stocked toolbox

Before embarking on an analysis of the use of ecosystem services valuation, it is in order to lay down the set of tools that will be the object of discussion.

As we said, the ESV toolbox is very well-stocked. These tools have been developed intensively for the last five decades and they are the object of an abundant literature. They come in many variants and as soon as one moves beyond presenting their basic principle, they become quite technical.

Fortunately, many very good systematic reviews have been proposed. Recent official reports on the use of valuation as a contribution to public decision-making (Chevassus-au-Louis *et al.,* 2009; UK National Ecosystem Assessment, 2011) include such reviews. The chapter dedicated to that review in the TEEB report (Sukdhev *et al.,* 2010) distils both the literature as a whole and such recent systematic reviews. We will then refer those readers who would like a closer look at the tools to that easily accessible literature.

For now, we will not go into any detail. We will provide here the minimum overview that is requested for the subsequent discussion, not of the tools themselves, but of their use. We believe indeed that, while tools are handled by their tradesmen – the economists – the results of their use, its consequences, intentional or not, are of much wider concern: to those who requested the intervention of the tradesmen, to those who are expected to benefit from it, to those who will be affected in any way.

1.1.1. *Characterising Ecosystem services: an integral part of ESV*

A first step of any ecosystem services valuation is to characterise the services that are provided by the ecosystem. The typology of such services is only gradually stabilising around categories like provisioning services (*e.g.* the possibility of extracting fibre or fish), regulating services (*e.g.* flood prevention), cultural (*e.g.* aesthetic enjoyment of landscape), or supporting (*i.e.* services rendered indirectly by supporting other services) (Hassan and Scholes, 2005). The difficulty is that our knowledge on such services is limited. We can rarely take for granted that characterising ES will be just a sort of technical preamble to valuation. It is a challenge in itself and it represents in fact an integral part of ESV (Fisher *et al.,* 2008).

One should not, however, exaggerate the difficulties. Ecosystem services are often presented as a new concept and perspective, as if we were at a pioneering stage, which is obviously not the case. Looking at the early literature on the valuation of natural assets in the 1960s it is quite clear that the bulk of what was being evaluated differed in name, rather than in content, from the ecosystems services under consideration today. For instance, early economic evaluators of wetland ecosystems (Sweet, 1971; Gosselink *et al.,* 1974; Nichols, 1983) were very clear that they based their work on what wetlands provide in terms of extractable resources, of water cycle qualitative and quantitative regulation, of amenities and of support for other ecosystem functions – in the current vocabulary, respectively provisioning, regulating, cultural and supporting ecosystem services.

The recent surge of literature and institutional attention on ecosystem services and their economic evaluation is neither a conceptual breakthrough nor a turnaround in perspective, but the consolidation and amplification of approaches that have been developed and promoted over the last five decades by ecology, environmental economics and environmental studies. We may need more knowledge as our demands for precision increase (for instance in terms of ES mapped location, of precise quantities, of exact thresholds), but what we already know would no doubt be enough for much better management of ecosystems than we are now able to organise.

1.1.2. *Assigning monetary value: different tools based on different logics*

Once ES are characterised, there are several possible approaches to assigning them a monetary value. Each of them is a different solution to the basic challenge of ESV: the fact that since there is usually no direct market of ecosystem services, one has to find a way to establish what their market value would be if there were one such market. So each approach proposes a way to indirectly assign market value to ecosystem services.

To summarise these approaches, we will refer to the well-known analytical framework of "Total Economic Value" as presented in most current environmental economics books (see for instance National Research Council, Heal *et al.* 2005), one of its first appearances being in Barbier *et al.* (1993).

The "Total Economic Value" framework is based on the idea that an environmental asset has an intrinsic overall value, calculated as the sum of use values and non-use values: "*Typically, use values involve some human interaction with the resource whereas non-use values do not.*" (Barbier *et al.,* 1993) p. 14).

Use values are then separated according to whether they are direct or indirect. Direct uses are where an individual makes use of a production or a service from the environment for his activity, be it a commercial activity, or a non-commercial one. Indirect uses refer to values that "*derive from supporting or protecting economic activities that have directly measurable values*" (*Ibid* p.15). This group of use values also comprises "option and quasi-option values", which refer to our potential future uses.

Lastly, non-use values are those values that are related to the mere existence of the environment, without intention of use, except when we value the environment to bequeath it to our offspring.

To exemplify these definitions, we will report here the table of values presented for wetlands by Barbier and colleagues).

Table 1 *Classification of total economic value for wetlands*

Use values			Non-use values
Direct use values	**Indirect use values**	**Option and quasi-option**	**Existence values**
• Fish • Agriculture • Fuelwood • Recreation • Transport • Wildlife harvesting • Peat/energy	• Nutrient retention • Flood control • Storm protection • Groundwater recharge • External ecosystem support • Micro-climatic stabilisation • Shoreline stabilisation, etc.	• Potential future uses (as per direct and indirect uses) • Future value of information	• Biodiversity • Culture, heritage • Bequest values

Source: Barbier et al., *1993*

Now, how are those values to be valuated?

To keep the example of wetlands running, the direct use values are calculated based on their market prices, for instance the value-added of beef cattle bred on wetlands per hectare of wetland, or the selling price of hunting rights, etc. Valuating indirect use is generally based on calculating the "avoided costs": costs that users avoid thanks to the support of services delivered: for instance, avoiding having to build dams, and/or dikes, and/or having to suffer damages from floods, etc. (Laurans and Cattan, 2000). Alternatively, it can be assessed by the travel cost methodology (inferring the value attributed to a qualitative difference of environment by the time and money spent to reach a satisfying site, *e.g.* for anglers). Lastly, to measure option and quasi-option values, the textbooks recommend using the contingent valuation of the agents' willingness to pay, which we will explain hereunder (Pearce and Turner, 1990).

Eventually, non-use valuation is based on replacing an absent economic demand by a simulated market, thus artificially re-creating a demand. This is generally done through enquiring about the amount of money people would be willing to pay to conserve the benefits, or would accept being paid in exchange for accepting to lose them. Such enquiries are done through contingent valuation surveys, or its variants like choice experiments. Hedonic pricing is another possibility (inferring the value of a difference in environmental quality from a difference of residential pricing, *mutatis mutandis*).

However, these various methodologies measure in fact different things, some disconnected, others, overlapping. For instance, contingent valuation is rather in itself a valuation of the total economic value (interviewees do not restrict themselves to a specific kind of values when responding to a survey), whereas choice experiments try to valuate separately the different sub-components of our willingness-to-pay.

Besides, the issue of how to frame accounting for ecosystem services is quite problematic, with difficulties such as establishing a comparison basis between a situation with and without a given service (TEEB 2009), the conditions for adding up the values of services of a quite different nature at different scales, and of avoiding double counting (Mace and Bateman, 2011).

Each of these tools, and especially direct use valuation and indirect use valuation, provides a way to combine ecological, technical, sociological or economic information to generate a monetary value for a given ecosystem service, or set of services.

A further set of tools is based on processing these "primary" values to produce "secondary" valuations of ecosystem services. Benefit transfer (or value transfer), for instance, consists in using values that were primarily established in some situation to value ecosystem services in another situation. It leads to the adoption of "reference values" of ES per ha of a given ecosystem type (Bergstrom and Stoll, 1993; Chevassus-au-Louis *et al.,* 2009). Costanza and colleagues applied this methodology when they assessed the value of the world's ecosystem services, based on unit values (calculated for specific examples, taken as mean references) multiplied by the surfaces of equivalent ecosystems (Costanza *et al.,* 1997).

Beyond this very rapid overview of the toolbox, let us now turn to our purpose, *i.e.* gauging the problems raised by the implementation and the use of those tools, rather than by their potential methodological merits.

1.1.3. *Ecosystem services valuations are context-dependent*

We mentioned above that a few literature surveys have addressed the problem of the Use of Ecosystem Services Valuation (OECD, 2002; Fischer *et al.,* 2008; Liu *et al.,* 2010; Laurans *et al.,* 2013). Those surveys have demonstrated that the various tools available are based on profoundly different methodological principles. They raise important theoretical and methodological issues which make their implementation usually far from easy and highly dependent on their context of use. For instance, data availability will determine the feasibility of using the tool (*e.g.* properly assessing the value of "flood protection" indirect use relies on the existence of pre-existing hydrological modelling). Another example is that the context will affect the circumstances in which valuation results will be used and thus discussed: a valuation tool based on inducing the value of recreational ecosystem services from touristic expenses will succeed more and be more useful in places where eco-tourism is an important component of the local economy. In the same way, assessing the benefits from pollination to crop farmers is somehow framing the problem as a flow of benefits from natural ecosystems to farms. According to the actors who use this value and their existing relations, this framing may be more or less relevant for decision-making.

Valuation tools thus tend to "frame" the problem in a given way. A concrete example of such a context dependence has been analysed by ourselves (Laurans, 2000; Laurans *et al.,* 2001), regarding the use of economic valuation of water policy benefits in France. At the end of the 1990 decade, French basin authorities were adopting their first River Basin Management Plans. The context was a dispute over the need to significantly upgrade the French water sanitation systems, in order to catch up the delay in implementing the European Commission directives in that matter, on the one hand, and the economic impact of those investments on household water bills on the other. The French ministry in charge of the Budget was worrying that this policy would further increase the rate of taxes and of water prices in the country, and wanted to question the rationality of these foreseen expenses, by putting their cost-benefit ratio into question. The water agencies, public bodies in charge of gathering funds and then financing the program, were supportive of these programs, which meant a significant rise of their budget and importance. The ministry in charge of Environment, sharing the final authority with the Budget ministry, was divided: some members advocated an upturn in water protection investments, whereas others regretted that the programs would lead mostly to building more sanitation systems, and would leave ecosystem maintenance and restoration at a minimum. All basin authorities, following the ministries' enquiries, had their benefit valuation studies carried out, but insisted that they should not be presented as "cost-benefit" assessments. They argued that no such tool was required to decide, as the program was mostly implementing EC directives and laws, thus already decided and not to be questioned. The Northern France Basin authority ("Artois-Picardie"), for instance, asked first that benefits of a future innovative water policy be valuated based on existing examples of such innovations. Three case studies were carried out on locations where ambitious programs had been implemented previously, and their important benefits were put forward as an illustration of what would come out of future comparable investments in the region. When presented to the assembly of representatives, who were to vote on the program, these results greatly pleased the environmentalist representatives. But they did not convince the local representative of Budget, who asked for a cost-benefit ratio to be calculated for the whole basin (not for selected positive illustrations), and for the program to be adopted (not just for examples of the policy issues at stake taken from investments). He asked for a systematic valuation, which would allow linking *ex ante* each possible level of investment with an amount of benefits (an "abacus"). A second valuation tried then to relate the cost of the program (only its part devoted to surface water and aquatic environment, including coastal) to its potential total benefits. The potential benefits were assessed mostly based on benefit transfers and on an extrapolation of indirect use benefits (gross). The resulting benefit-cost ratio was small (less than 0.5). However, this result did not deter support and votes for the program, which was largely adopted. Industry representatives, who were to pay a significant share of the costs, did not take argument of the weak benefit-cost ratio to refuse their vote. Conversely, they expressed their satisfaction that this result was demonstrating, to the others and especially to the environmentalists, the importance of the efforts they were accepting, in some sort of "advertising effect".

This example illustrates the variety of choices that usually have to be made about ESV methods in a given context of decision. Intentions that sustain environmental valuation are usually heterogeneous: arguing for or against an investment program, demonstrating the economic benefits that result from an innovative policy, or the importance of costs that a category is currently accepting to bear, etc. According to these intentions, the requirements and qualities of the valuation tools are not equivalent. For instance, local examples and detailed case studies, based on past programs, were appreciated by some for their careful links between hydrological, ecological, social and economic data and reasoning; they were also geographically rooted and referring to concrete locations, communities, equipment etc. These qualities were effective for those who thought that the program was mandatory by law, and was only to be justified in the eyes of local actors in order to facilitate its adoption and further implementation. But this was not the intention of some others who wanted either to put forward a precise estimate of their future costs or to try and adjust the program's content and level of funding according to the expenses that would produce the highest benefit-cost ratio.

In brief, valuation tools are not simple, straightforward, uncontroversial tools. They are sophisticated or simple, but always imperfect; they may be detailed but still remain controversial. As the review of the valuation toolbox in the TEEB report (ten Brink, 2011, p.242) concludes: "*it should become clear that techniques to place a monetary value on biodiversity and ecosystem services are fraught with complications, only some of which currently can be addressed. They lead to results, the exact reach and meaning of which requires expert discussion, a discussion that is made difficult by the complexity of the tools and their implementation. They require that the evaluators master a range of skills, from ecology to economics, and that they constantly exercise judgement to fit method to context.*"

In other words: the ESV toolbox is well-stocked, but requires skilled tradesmen and educated clients for its use. It may produce useful results, but that is in no way easy and guaranteed, since it will depend on skilled implementation, and on in-depth debate and judgement as to the relevance of results to the problem at hand and the decision-making context.

These basic facts about ESV's dependence of their context of implementation are to be kept in mind when discussing the role and use of ESV for biodiversity conservation. We should also be aware of the different critiques made on the very principle of ESV, and pay attention to how ESV promoters and critics have been interacting up to now.

1.2. A barrage of critiques on the principle of ESV

As ESV was developed over the last four decades, it was accompanied by a barrage of critiques against the very principle of attributing monetary value to ecosystems and their services.

1.2.1. *Attacks from all parts against reducing the value of ecosystems to services and economics*

Critiques against ESV came from all quarters of the environmental field.

First of all, they came from conservationists and conservation biologists: "*there is a range of problems associated with valuation of ecosystem services*" (Redford and Adams, 2009, p.787). These critiques have more particularly expressed three major concerns about monetary valuation. First, because of all the simplification it implies and of the salience it gives to the more directly utilitarian aspects of ecosystems, it risks missing essential aspects of ecosystems' functions and value. Those aspects may be connected only more indirectly to services that we use, but they are nevertheless fundamental to ecological balance and richness. "*Not all ecosystem processes sustain and fulfil human life. Processes such as fire, drought, disease, or flood work against this goal, yet they are vital for ecosystem function, structuring landscapes, and providing vital services and regulatory functions to nonhumans*" (Redford and Adams, *ibid*). Second, monetary valuation tends to compromise the ethical, moral and aesthetic standpoints of conservation, either because it retains only the more utilitarian aspects of ecology, or because it expresses non-utilitarian aspects in the language of interests (for instance through concepts such as "existence value"). Last, it may become counterproductive in terms of conservation strategies if the causes defended by the conservationists have more strength on the basis of other values than on economic ground alone: "*economic arguments about services valued by humans will overwrite and outweigh noneconomic justifications for conservation*". (Redford and Adams, *ibid*).

Social scientists and philosophers have also been actively involved in this process of critiques. They have notably voiced concerns that monetary valuation tends to discount all worldviews that are not utilitarian, or force these views into expressing themselves in such a framework (Clark *et al.*, 2000). In their view, monetary valuation tends to push aside the culture, the knowledge, the views of local people, who are the very people most involved in the ecosystem's day to day management, often poor people whose dependence on the ecosystem's services are particularly vital.

It also risks short-cutting the political life and the political decision-making process should cost-benefit calculations become the main rationale of public decision-making (Martinez-Allier 1987; Spash, 2008).

Finally, more critical economists attack ecosystem services valuations because the welfare economics framework they rest on is based on too narrow a view of the economy, and cannot appropriately encompass the real scope of the biodiversity crisis and of its potential social consequences (Georgescu-Roegen 1986; Costanza and Daly, 1992; Daly, 1992); reviewed in (Gomez-Baggethun *et al.,* 2010).

1.2.2. *Beyond the sheer confrontation between ESV promoters and critics*

We have just listed here a few themes of the critique of ESV, but they easily fill entire books (see for instance (O'Neill, 2007) or (Fullbrook, 2004)). It would be only a slight exaggeration to state that the critique of the principles of ESV has become a research field in its own right!

A stereotyped view of the confrontation would show on one side an environmental economist insisting that public decisions be made on the basis of rational cost-benefit reasoning, minimising politics and their inconsistencies, and on the other side a critic attacking this not as an effort of rationalisation but of imposition of market reasoning at the expense of all other ecological, social, political and ethical considerations.

However, as one reads the recent literature, a different picture emerges. An extreme awareness of critical arguments is apparent throughout. This is especially true for non-academic literature, and the various recent reports promoting ESV, be it the Center for Strategic Analysis (CAS) report in France (Chevassus-au-Louis *et al.,* 2009), the UK National Ecosystem Assessment (2011) or TEEB (Sukhdev, Wittmer *et al.* 2010). In ESV assessment and promotion reports, the critical literature is cited and discussed, at least to a point (see for example Chevassus-au-Louis *et al., ibid,* 142-143). The theoretical difficulties and limits of the methods are readily acknowledged and discussed (see for example TEEB, chapter on economics of ESV). It is repeatedly underlined that ESV is only one input into the decision-making process, that it has to be combined with other inputs and that the process remains and ought to remain political (Arrow, Cropper *et al.,* 1996). Limits are set to the relevant scope of situations where ESV is claimed to be useful; for instance the CAS report limits the relevance of monetary valuation to "ordinary biodiversity", considering that in "exceptional biodiversity", aesthetic, ethical and social values are so essential that one should not run the risk that valuation may suggest that destruction of biodiversity is a reasonable option based on its economics.

Rather than the stereotypical confrontation of the economist and the social scientist, the current state of play seems to us to be reflected better by the following quotations from the TEEB report: "*valuation mechanisms should be seen as part of a broader range of diagnostic and assessment tools and political-institutional mechanisms that facilitate the understanding of complex socio-ecological systems" (p. 175); "despite these limitations, demonstrating the approximate contribution of ecosystems to the economy remains urgently needed* [...]. *Valuation exercises can still provide information that is an indispensable component of environmental policy in general. Ignoring information from valuation methods is thus neither a realistic nor a desirable option. Instead, policy makers should interpret and utilize the valuable information provided by these techniques while acknowledging the limitations of this information* (ten Brinck, 2011, p. 242)"; This standpoint is particularly interesting for us on two counts. First, it expresses the effort to integrate the critiques into advice about the practice and use of ESV. Second, it indirectly points to the new agenda implied in the current state of practice and debate of ESV: focusing attention on use, use contexts, and the actors involved both in ecosystem management and the use of ESV.

However, even if the academic sphere as well as experts and policy advisers such as the TEEB project members increasingly acknowledge that ESV is far away from "ready-made" and standard instruments, the prevailing change of language and tone we just described does not necessarily signal an end to the tension between promoters and critiques of ESV. On the side of the first, not all experts and enthusiasts of ESV embrace the same moderation and prudence about what the tools can and cannot do, nor what they should or should not be used for; recall for instance the words of Freeman III, only a few years ago: "*Once the objective of maximum net economic value or economic efficiency has been accepted, policy becomes an almost mechanical (but not necessarily easy) process of working out estimates of marginal benefit and marginal cost curves and seeking their point of intersection.*" (Freeman III, 2003, p. 10), p. 10."

On the side of the critics, some rightly point to the fact that important and deep tensions are inherent in the very principle of assessing ecosystems and their services on a monetary plane (O'Neill, 2007) – tensions and contradictions that will not be suppressed by any effort to integrate other aspects of environmental issues into monetary values.

Besides, even if economists are well aware that real-life decision-making is not a mechanical process, ESV is rooted in a social choice perspective, where social optimum is the goal, and valuation is a measurement made to integrate environmental benefits or costs in calculating that optimum. ESV authors, as TEEB exemplifies, strive to adapt ESV to the more complex nature of decision-making. The methodology and the basic principles of ESV, however, remain marked by their theoretical, social choice origin.

The debate between proponents and critiques of ESV, then, is not over. Yet overcoming the current lukewarm stalemate and becoming fruitful again means moving resolutely to a new ground. In our view, much of this new ground lies in putting forward much more explicit and precise analysis – and construction – of the wider contexts of ecosystem decision-making, management and policy.

These contexts are indeed decisive both to proponents and critics of ESV. To proponents, because "*the chosen methodology of valuation depends on the purpose of valuation*" (Sukhdev, 2011, p.XXIV) and the purpose can be appreciated effectively only on the background of an adequate understanding of the context of decision-making and collective action. To critics, because assessing the risks involved in monetarising ecological services requires analysing and debating the actual consequences of using ESV, on political, social and economic, as well as cultural grounds, and the contexts within which these concerns materialise – or not – on the ground.

Therefore, we will first look at what kind of uses the literature sees for ESV *in principle*, then at how it deals with the *actual* use of ESVs on the ground. Based on the results, we will see that difficulties in the use of ESV for decision-making have been a long standing problem, are still a problem today, and we will look at some directions for understanding and treating these difficulties.

1.3. Uses of ESV: the expectations

In a recent review of ESV literature , we have found that only a small proportion of papers on ESV focus on its uses in decision-making and action (Laurans, Rankovic *et al.* forthcoming). We identified, however, nine references that propose reviews of what these uses should be [5]. Throughout these nine references, and then through

[5] Navrud and Pruckner, 1997; Pearce and Seccombe-Hett, 2000; OECD, 2001; OECD, 2002; Turner *et al.,* 2003; National Research Counci *et al.,* 2005; Secretariat of the Convention on Biological Diversity, 2007; Fisher, Turner *et al.,* 2008; Liu, Costanza *et al.,* 2010.

the more than 400 papers we reviewed, the use of ESV (UESV) is envisaged in three basically different ways. These three categories are of course not necessarily exclusive, as ESV authors can suggest that their results be useful for different purposes. However, most ESV texts refer – and most often, just cursorily – to just one of these three categories: they see ESV as a candidate for a *decisive* use, a *technical* use, or an *informative* use.

1.3.1. *"Decisive" ESV*

Many authors refer to the use of their ESVs as a means to contribute to a process in which a given choice is to be made, ex ante, by a decision-maker facing various alternatives. ESV is to be incorporated into a cost-benefit analysis (CBA), to provide elements on the opportunity of a project and its economic consequences with regard to ecosystem services, thus enabling choice.

Within this category, three variants of UESV can be distinguished.

ESV for trade-offs. By proposing a monetary value for ecosystem services, it can be expected from ESV to factor environmental concerns into the cost-benefit analyses (CBA) that are underpinning decision-makers' arbitrations. In this respect, the purpose of the ESV is to enable the decision-maker to optimise social well-being by making a choice that balances out the different preference criteria.

Participative ESV. Instead of providing a comprehensive range of choices that reflect a socially optimal decision, ESV can also be seen as a basis for discussion: through an open debate on ESV parameters and assumptions, stakeholders negotiate and define a project that is adjusted and enhanced in terms of compromise and the sum of interests. In this perspective, we are still in a configuration where ESV is potentially 'decisive', and where it intervenes *a priori* as a key decision-making tool.

ESV as a criterion for environmental management. Within limited budgets targeting environmental objectives, ESV can also be considered as helping allocate conservation efforts within an organization, in an optimal way. ESV as a management criterion differs from arbitration in that it concerns only a specific organisation that has to make choices for its own actions, and does not entail a choice among wide policy and social priorities. But it still remains a tool that is expected to be "decisive" in its weighing of alternative courses of action.

1.3.2. *"Technical" ESV: fine-tuning an economic instrument*

This second category involves those cases where ESV is applied to adjust an economic instrument that has been chosen to implement a given decision. It covers two possible types of UESV.

ESV for establishing levels of compensation. Agents responsible for ecosystem degradation can be obliged to pay compensation for such damage. This compensation may be *a priori* (*i.e.* compensating the anticipated effect of an operation), or *a posteriori* (*i.e.* remediating some of the damages already caused by an accident). In both of these cases, valuation is expected to provide guidance for administrative decisions or court rulings that determine the amounts to be paid out (OCDE, 2002).

ESV for price-setting. In other cases where an economic instrument has been chosen, ESV can be promoted to determine the amounts payable on the basis of a willingness-to-pay or willingness-to-receive logic: payments made by the beneficiaries of services in the case of Payments for Environmental Services, entrance fees to nature parks, etc. ESV can also be suggested to fix prices that allow externalities to be internalised, for example by factoring environmental costs into the price of a product.

1.3.3. *"Informative" valuations for decision-making in general*

Aside from its decisive and technical role, ESV can also be viewed as a means of providing information intended to have an indirect influence on decision-making, considered in a very broad sense. Three variants of this category of UESV are proposed in the literature.

ESV for awareness-raising. Informative ESV may be seen as the vector for a broad-based message concerning the preferences that should be mainstreamed into society, particularly to ensure that ecosystem services considerations are integrated into public choice.

ESV for advocacy. In this variant, informative ESV addresses a specific decision or an identified type of decision, as it is the case for the 'decisive' ESV category. However, whereas for 'decisive' valuations ESV is deemed neutral and should inform an optimal choice, here it is more a matter of showing that an already identified choice is justified, either *a priori,* to show the economic relevance of the measures envisaged, or *a posteriori*, in which case ESV may serve as a tool for verification. Pearec (2000) and Daily *et al.* (2009) for example, basically consider that any ESV is a form of 'advocacy'.

ESV for producing 'accounting indicators'. This last variant involves situations where valuation is designed to allow decision-makers, or public opinion, to stay informed on the state of the natural capital and potentially to integrate this information into their decisions in general. This category encompasses the ESVs which aim at building natural heritage accounts.

These three main categories, and the number of their occurrences in a selection of 433 texts, (mostly peer-reviewed, but some of grey literature type), are displayed in Table 2. The table also distinguishes texts where the potential UESV is only cursorily referred to (generally by one or two sentences in the conclusion or in the abstract), those which address a specific UESV as a subject, at least in some part of the text, and those that document a situation where ESV has been used.

Table 2 *Expected uses of ESV in a selection of 433 references about ESV*

	Cursory reference	Analyse of the use issue	Documentation of use cases	Total
Trade-offs	113	15	8	136
Participative	14	1	3	18
Environmental Management criterion	36	0	0	36
Compensation level	10	1	2	13
Price setting	23	0	1	24
Awareness raising	79	4	1	84
Justification	96	2	0	98
Indicators (accounting)	20	4	0	24
Total	391	27	15	433
	90%	6%	3%	

Source: Laurans et al., *2013.*

Laurans *et al.*'s review shows that a fairly clear picture now emerges from the literature on what uses for decision-making could or should be expected from ESV. It clearly appears that these expected uses can be very different from one another, which leads to three considerations we think important to underline here.

The first is that the choice of methods for ESV and the way they are implemented can and must be guided by a precise definition of the sort of use that is expected. This point is already often made in the literature. It is nevertheless often overlooked in discussions of ESV. But most of all, to be translated into action, it requires appropriate tools for matching methods, contexts and uses. This is precisely one of the areas where progress is necessary, and can be expected from more explicit treatment of use issues.

A second consequence of the diversity of expected uses is the need to discuss the consequences of using ESV in full view of specific expected uses, rather than criticising or promoting them through stereotypes of use (where for instance economic valuation would simply replace political decisions). For instance, the level of risk that ESV may lead to shortcutting political views is very different whether it is designed for a "decisive" or an "informational" use. Consider the following case. In the context of coral reefs regions, if one wants to proceed to a valuation of coral reefs ecosystems services so as to integrate them in a CBA of projects that would determine decisions, this would be inappropriate. Even leaving aside the principle of CBA-based decision, available ESV methodologies have not yet reached the robustness and accuracy that would be needed for that kind of use (Laurans *et al.,* 2013). Many methodological difficulties prevent them from adequately reflecting the relations between a coral reef condition and its use as well as non-use values: the fishing capacity is not easily expressed in such specific ecosystems, and the "maximum sustainable yield", extensively used in other parts of the world, is not easily applicable there (David *et al.,* 2007; Pascal, 2010). Non-use values attributed to the reef by their traditional inhabitants are not built on the same basis as would be those of a Western resident or tourist (O'Garra, 2009). But, on the other hand, the very same studies that would be inadequate for weighing options for decision can be very useful if they are considered locally to raise awareness about the importance of such ecosystems for the local economies, and about the need to question the most harmful current policies (overfishing, gravel extraction, water pollution…).

A last remark is that the various expected uses listed here relate very differently to the economic theory. Some of them – ESV for trade-offs, as a criterion for environmental management, or for price-setting – are related to the economist's paradigm of decision-making as the search for a collective optimum. Other uses take much more distance (but never to the point of being completely disconnected) from economically optimal decision-making: ESV as language for participatory decision-making, for awareness-raising or for justification and support, for instance, do not pre-judge about what will be the organising logic of decision-making. As a consequence, the place of economic theory in the framing of discussion of ESV use should be envisaged in a different way, depending on exactly the type of use that is under discussion.

1.4. Actual use of ESV for decision-making: a blind spot in the literature

If we now turn from expected to actual uses of ESV for decision-making, the picture that emerges from the literature is indeed quite different.

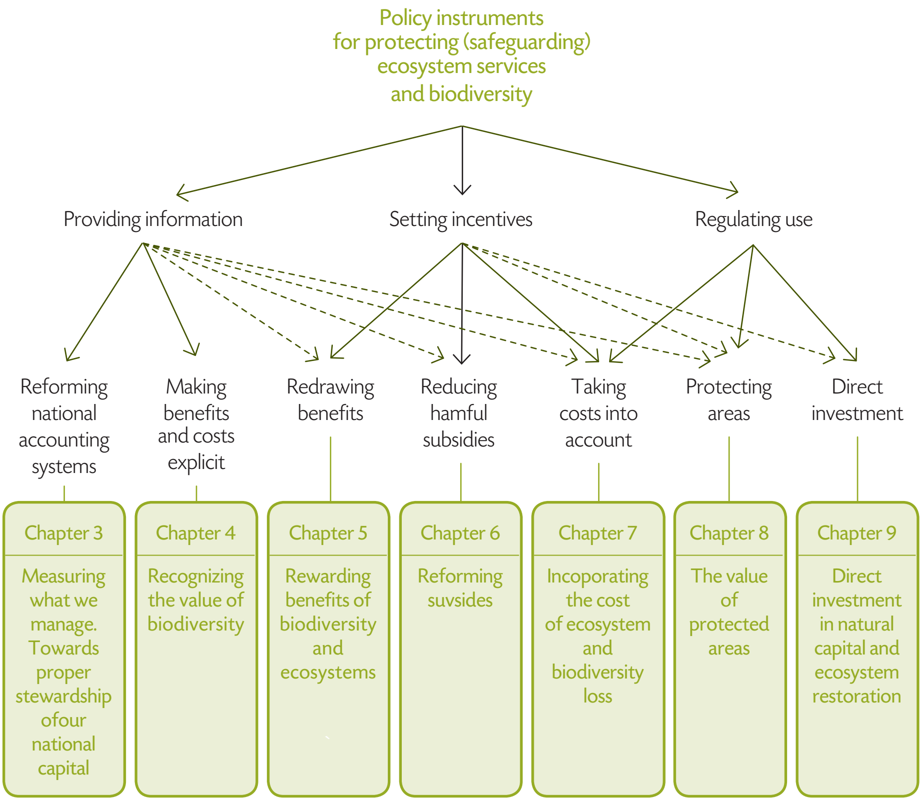
1.4.1. *Most of the expected uses are really hard to find in practice*

As far as "decisive" use of ESV is concerned, it is remarkable to note that the ESV literature, however profuse about methodology or valuation cases, seems almost silent about real-life cases where an ESV was commissioned to allow a choice. Out of 190 analysed references that suggest that ESV could be used for decisive purposes, we only found 11 (around 6%) reported cases where valuation has actually enabled decision-makers to choose a policy, a program or a law (cf. Table 2. Expected uses of ESV in a selection of 433 references about ESV (Laurans *et al.*, 2013). This tends to confirm the OECD's statement that "*although fairly common in the environmental economics literature, valuation techniques have remained somewhat peripheral to environmental policymaking on major issues*" (OECD, 2002, p.18).

Expectations regarding "technical use" also seem far from being fulfilled: we found only 3 references documenting effective use of ESV as technical for a total of 37 references mentioning this category of use (cf. Table 2). As Liu *et al.* (2010) put it: *'Indeed, one would imagine that ESV, the process of assessing the benefits of environmental services, must have been applied widely to guide payments for ecosystem services… In practice, however, ESV results have rarely been applied in setting payment amounts'* (p. 2068). This rarity of use of ESV in PES programs is confirmed by the PES literature (Wunder, 2006).

Reference to a potential use, in our literature selection, is often of the "informative" type (47% of the total 433 references refer to advocacy, justification, visibility of biodiversity on the political agenda, etc.). This is also well reflected in the TEEB policy options overview diagram (See Figure 1), in which EVS implicitly falls entirely into the category of tools "providing information". But even though this type of ESV use is so central, this fact is hardly discussed in the ESV literature: only a small number of the papers reviewed by Laurans *et al.*, (2013) have considered utilisation of ESV as a subject of attention (15 references) and only one reference discusses an effective informative use of ESV.

Figure 1 *Contribution of ESV to decision-making according to TEEB*



Source: ten Brinck, 2011.

The result of this review concurs with other recent reviews (Fisher, Turner *et al.*, 2008; Liu, Costanza *et al.*, 2010) in finding the subject of effective use disappointingly addressed in the literature.

1.4.2. *ESV implementation: poorly documented, or actually poor?*

The next question to turn to is: what are the possible explanations for this result? Two drastically different hypotheses may be summoned here.

The first is that ESV are indeed used but that this use is under-documented in the literature, especially in the environmental economics literature. There could be several plausible reasons for such a bias. Use of ESV could be difficult to observe. Calculating values – despite the many difficulties it involves – is one thing; but following how they are used in, and how they affect decision-making processes that are highly complex, is quite another. Distinguishing what serves to make and what serves to justify a decision is often a daunting challenge. It may also be that use of ESV is not yet, or has only recently arisen, in the research agenda of the field. Or, it may be a research issue, but not for economists, falling rather in the purview of social scientists interested in how decisions are made, which would explain that it appears as a blind spot in the environmental economics literature. A last potential explanation is that it may fall almost completely out of the academic domain, and be only a practical issue, explaining its relative absence in the mostly academic literature on the subject.

The second hypothesis is that the rarity of references to use reflects the fact that the level of actual use of ESV is indeed small, compared to the place ESV occupies in the environmental economics (and the more general environmental policy) literature. The environmental economics literature advances several possible causes that may explain this non-use, if it were to be confirmed. ESV may not be accurate enough, especially for decisive use (see above, our example about coral reefs ESV). Or it may lack relevance, for instance because of discrepancies between what the valuation measures, and the issues that are actually at the heart of decision-making processes. The high cost of ESV is another often mentioned possible explanation, especially when ESV is seen as more relevant for local decision-making, where the cost of valuation may easily be felt to be disproportionate to the stakes. Three remaining possibilities locate the cause for disappointing levels of use not primarily in issues with valuation, but in the characteristics of the decision-making context. Some environmental economists feel that decision-makers do not have enough understanding of valuation to understand ESV or to trust it to the point of using it. Or legal and regulatory frameworks for decision-making may be an obstacle to the use of valuation. Or, finally, ESV may clash with the politics of decision-making. As Robert Hahn (2000, p. 18) puts it: "*Politics affect the process in many ways that can block outcomes that would result in higher levels of economic welfare. [...] Policy ideas can affect interest group positions directly, which can then affect the positions of key decision makers (such as elected officials and civil servants), who then structure policies through the passage of laws and regulations that meet their political objectives*".

1.4.3. *Three avenues for progress in response to the inadequate use of ESV*

In response to the second hypothesis – an actual deficit in ESV use – three avenues for progress are proposed by the literature : (1) to invest in improvements of ESV techniques so as to make them fitter for use (Fisher *et al.,* 2008) or/and less costly (Loomis and White, 1996); (2) to change the contexts of use so as to create conditions that would give rigorous valuation a bigger role in decision-making (training decision makers, modifying legal framework, etc.); (3) to accept that decision-making, management, policy and politics have their own logics and legitimacy and that the priority should be to better understand their own, complex logics to understand how ESV could have a stronger impact on them. It is more precisely to this latter challenge that this book intends to contribute. It addresses the need of ESV researchers "*to understand how the political process affects outcomes and [to] actively market the use of appropriate and feasible methodologies for promoting environmental policy*." (Liu *et al.,* 2010, p. 73).

1.5. Improving ESV: a focus on content, or on process?

But as ES valuators embark on this effort of connecting with the wider context of decision-making, it becomes apparent that they do not all share the same concept of what valuation essentially consists in. To approach these differences in perspective, which are pervasive in the literature and quite manifest in our interviews, we will follow Godard and Laurans' analysis (2004). They show that the field is polarised between two very different views of the nature of valuation.

1.5.1. *Two contrasting approaches to ESV: revealing value, constructing value*

They call the first approach "value revealing approach" (VRA). In that perspective, value derives from individual preferences that are independent of the collective decision-making process. The core aim of valuation is then to measure these individual preferences and to aggregate them in ways that reveal what collective preferences should be. If one assumes policy-makers should have as their main goal to maximise economic welfare, then these valuations should orient decision-making as directly as possible. If one has a more political view of decision-making, then the VRA approach provides as one input a clear, legible measurement of the impacts on welfare of various possible courses of action.

By contrast the "social process approach" (SPA) is based on the view that value is based in large part on conventions rooted in collective action frameworks and in interaction between actors. Here, rather than the objective measurement of pre-existing preferences, valuation involves constructing individual and collective preferences that are partly produced by the decision-making situation itself. In that perspective, valuation methods cannot be seen as independent of the decision-making situation and process, but are at least in part grounded in them.

Whether one adopts one or the other perspective radically affects the way the context of ESV is perceived and should be dealt with.

Table 3 *Status of environmental valuation*

	Environment valuation as a 'social process' (SPA)	**Environment valuation as a value-revealing process (VRA°**
Status given to the *actual collective decision process*	Object for knowledge Relevant for valuation, and for the meaning and use of results	Irrelevant object Actual decision mechanisms are viewed as imperfect or irrelevant versions of an ideal concept of collective decision mechanisms
Status given to *evaluators*	Actors and producers in a social process Supposed to provide appropriate information according to legitimacy frameworks, and to find compromises between personal attributes and social roles	A technical instrument Ideally, individual attributes of evaluators are of no significance
Status given to *people's preferences*	A social construct Preferences are not fixed as essential characteristics of individuals but framed by social interaction. Answers and data are co-produced by the subject and the resource-people	Object for knowledge Individual's preferences are real, and pre-existing. Positivism.

Source: adapted from Godard and Laurans, 2004.

In a VRA approach, the focus of attention is the internal validity of ESV. The evaluators' judgement is involved only inasmuch as it guides them to produce a valuation that is technically as good as possible. The decision-making context and its many variants are not *per se* an object of interest. Navrud and Pruckner (1997) conclude their report on the use of ESV in EU decision-making by stating that "*increased cooperation of scientists and economists, who would assess risks and weigh costs and benefits, is called for to inject greater rationality and cost effectiveness into environmental rule-making*".

In SPA, the evaluator is one of the actors in the decision-making process. Not only his theoretical and methodological resources, but also his judgement and relations with other actors, are involved in valuation. There is no objection here to different evaluators reaching different values due to different choices of methods and assumptions based on different framings of the context. Understanding, interpreting, analysing the decision-making process itself becomes very important because it deeply affects the valuation and its meaning.

1.5.2. *An opposition that should be neither over- nor under-played*

To anyone familiar with environmental and/or ecological economics, this dichotomy will sound familiar. We want to underline here that the opposition between the two views should neither be played down nor up too much.

It should not be played down because it is true that in principle, the two views are opposed in almost all respects, and in very fundamental ways. Where one view would recommend that the valuators become ever more detached from the decision-making situation, the other would have them be more and more sensitive to it; where one view would frown on the valuator manipulating the agents' preferences, this is precisely what the other recommends, through various participatory methods for instance.

But if one plays up the difference too much, one misses common ground that is essential to the field.

First, in practice, there is much exercise of judgement, even if it remains largely implicit, on the part of the best "VRA oriented" valuators. Symmetrically, there must be a substantial input of objectifying values in a "SPA oriented" valuation, lest contributing to the process as an economist would hardly be different from contributing as a facilitator of a participatory process. So each perspective chooses to insist on opposite roles of valuation; but as much as there are important differences, there is also much common ground (in the repertoire of methods, for instance, as well as in practice on the ground) that it is essential not to miss.

Second, it is important to realise that from the standpoint of the utility of ESV for environmental collective action and decision-making, each approach makes useful contributions. VRA provides an external, critical view on values, costs, benefits, efficiency; it allows judgmental analysis of public policies, based on criteria that are not easy to manipulate. SPA builds into ESV bridges between economic language and methods on the one hand and social and political contexts and processes on the other hand. It actively accommodates interdisciplinary work at the intersection between economics and environmental studies.

1.5.3. *ESV as a value revealing approach: understanding contexts and dynamics of use is necessary if results are to impact decisions*

Starting with VRA, one may think at first that such approaches would have no need for much understanding of the context of their possible uses. Indeed, since valuation, in that perspective, is an objective measurement and aggregation of preferences, there is no fundamental need to know what the measurements are going to be used for. There might even be a risk involved of the process tainting the valuation. In the terms of Pearce and Seccombe-Hett (2000): "*while a preoccupation with the process is understandable, one aim of valuation is to provide a check on the efficiency of decision, however they are made.*" In this "balancing role" played by valuation, a certain indifference to how decisions are made may be quite useful.

But this does not imply that VRA should be completely indifferent to process. To discuss this, we can follow up on the phrase we quoted above (Navrud and Pruckner, 1997): the role of valuation is to "inject greater rationality and cost-effectiveness" into decisions. This choice of words reflects quite well the neat separation between the production of the medicine (valuation) and the process of introducing it into decision-making process to make a difference. But the separation should not be complete. To extend the metaphor, one could hardly condone injecting a drug to cure a patient (here, from inferior rationality) without medical knowledge and nursing know-how. The knowledge required involves understanding enough of the processes at play to be able to assess how the injection may have the capacity to improve the situation; it also must include an understanding of the forms, excipient, injection route and doses in which the drug may best be taken up by the processes that need improvement. As for the nursing know-how, it also takes applied skill to administer some drugs. And so it goes as well for VRAs: a rational critique of decisions and of cost-effectiveness, for instance, may sting considerably and the political process may not welcome the injection at all…

To sum up, understanding how decisions are made can be useful for VRA if it allows identifying decision-making processes that are contested by the decision-makers and/or the other actors involved. Then "injecting some rationality" may be useful but an important issue, however, is to identify who voices the complaint. The case where decision-makers are aware that they have a problem and seek guidance is straightforward. Conversely, the major critique of VRA corresponds to cases when economists think decision-makers and actors have a problem, missing, for instance, opportunities to maximise economic welfare as was suggested in Hahn's quotation cited above (2000). Yet, decision-makers tend to consider they are fine and have every right to make decisions based on the political processes and criteria they like to choose (social, cultural, etc.). Insisting on injections in such cases is really objectionable, a point made by Sagoff (2011, p. 501) when he objects to the interest of ecosystem services valuation: "*Market actors, interest groups, and property owners seem to have a good handle on the ecosystem services that affect them and they do fairly well in bargaining with each other to manage conflicts and scarcities. It is not clear that any stakeholder or user group would have acted in an economically more efficient manner in the light of 'a new conceptual model for the interactions among service providers, supporting systems, service provision, and societal and environmental changes'* (...)".

A key role of environmental economists is to introduce, in the decision-making process, information about ecosystem services and their values that are overlooked. But it is essential that the patient's consent be obtained. A crucial part of ESV's success is based on its capacity to sustain a dialogue between valuators and actors, with respect to the exact nature of the problem. Besides, this ability is the main thread of most recent official reports on ESV. Otherwise, without the patient's consent, economists should prepare to justify their intervention and sustain vigorous resistance from their "unwilling" patients.

In both cases, the main question that arises is developing a finer understanding of the reception of economic rationality messages and of the reaction to them. For instance: if parts of society (environmental NGOs and a part of the public) are quite clear that there is a problem, but others (for instance, industry representatives) deny there is one, how should the situation be interpreted, and how should one proceed? Or if governments issue official statements to the effect that they acknowledge a serious problem and that significant action must be taken in the not too distant future, does this signal that the "injection" has succeeded?

The other part of meaningful work regards the "nursing" in our metaphor: what forms of ESV information, communicated to whom and in what form, can guarantee that actors of the decision-making process receive valuation when they need it, timely and usable so as to make a difference?

In straighter terms, following up how valuation affects decision-making and when and how it contributes effectively to taking charge of ecosystem services should form a first axis for analysing how ESV, when conceived as a value-revealing instrument, is relevant for decision-making.

Addressing those issues seriously requires an in-depth understanding and detailed analysis of decision-making processes.

1.5.4. *Social process approaches to ESV: understanding contexts of use is a prerequisite for the evaluation itself*

When considering ESV as a social process, there is also a need for understanding decision-making processes and management contexts, but for different reasons. Firstly, in SPA, the separation between valuation and decision is itself called into question, as an understanding of the decision-making context becomes an integral part of constructing priorities and values. Therefore, in that case there is no doubt that explicit and relevant models of how decisions about biodiversity are made, and how bifurcations between possible courses of action are played out is a fundamental resource for improving ESV. Secondly, articulating social processes, methodology and data is needed in SPA. This articulation is the strength, but may also be the Achilles' heel, of SPA approaches. They require that the valuators master the repertoire of economic valuation methods, and that they should be able to acquire and treat data in a coherent, but also adaptable way. Their resources for analysing how elements of valuation combine with particular configurations in the decision-making process become then a major focus of their expertise.

To conclude this discussion of VRA and SPA approaches, focusing on the use of valuations (rather than on their foundation and method) does not dispel the deep differences in perspectives that fuel controversies in environmental and ecological economics. But it also does not bring one back to a caricatural opposition. Resources for a better understanding of the decision-making process and biodiversity management contexts can help VRA valuators assess where, how and in what form the information they produce can make a difference for biodiversity management. As for SPA valuators, although they are highly motivated to integrate such understanding into their valuation frameworks and methods at all stages, their expertise remains rooted in economics, and they still have considerable needs for complementing it with external resources for analysis of decision-making, management and policy.

1.6. Teachings from a series of thirty-year-old case studies

Overall, we are advocating here ESV studies that combine on the one hand a detailed understanding of the decision-making context and process involved, and on the other hand, a valuation that is both rigorous in terms of its economics, and sensitive to context. To illustrate the benefits of such a balance, let us turn to results that came out of a thirty-year-old research project.

1.6.1. *Are decisions that impact biodiversity really taken on the basis of economic reasoning? Discussion in the 1980's*

In 1980, the French Ministry of Environment asked the renowned environmental economist Claude Henry to lead a research program to "design a method for evaluation of environmental consequences of infrastructure projects on the environment" [6]. The rationale for the Ministry to extend the research budget sounds familiar: the officials commissioning the study thought that to take environmental impacts into account in decisions on infrastructure projects, these impacts needed to be measured up to the economic advantages that motivate such projects and that if valuation methods were available, this could prove decisive in tilting a balance of decision-making that was all too often unfavourable to environmental concerns. On the part of the Environment Ministry officials who commissioned the study, there was no naïve vision of perfect decision makers that would balance in their purified rational minds the pros and cons of projects and reach an optimal decision. They were all too aware that such decisions were made in the Prime Minister's offices, following a confrontation of views and of files between the counsellors of the Ministry that wanted the biodiversity-damaging project and advisers for the Environment Ministry. What the latter sought in the development of ESV was arguments that would succeed more often in obtaining an arbitrage that would stop or at least modify some projects with excessive negative consequences on the environment. Since such projects were routinely advocated by the sector-based ministries promoting new infrastructures and development projects in terms of economics and employment, valuing environ-

[6] Development of a method to evaluate the environmental consequences of large development projects; SCORE/Laboratoire d'Econométrie de l'Ecole Polytechnique/Environment Ministry, 1982/

mental impacts in economic terms, so as to be able to balance economic benefits expected from projects with the economics of their negative environmental impacts, seemed a good idea.

Before embarking on research on the methodology for monetary valuation of impacts on ecosystems, however, Claude Henry raised a preliminary, fundamental issue. He insisted that for the whole enterprise to be relevant, one had to make sure that decisions on infrastructure projects (with biodiversity impacts) were indeed taken on the basis of cost-benefit data and reasoning. To answer that question, he launched a series of four case studies on recent or ongoing decisions on infrastructure projects in France: a hydroelectric dam, a new motorway, a large-scale drainage scheme and an industrial waste dump site. On each project, the decision-making process was studied in minute detail, including (but not disproportionately focusing on) the way economic studies were used in the process. The contents of such economic studies as were present were analysed and criticised in detail.

In all four cases, some form of economic cost-benefit-type study had been provided by the project promoters. In none of the four cases could it be reasonably argued that they had been decisive in the decision-making. In two cases – the drainage scheme and the industrial waste storage facility – the studies were hardly used in discussion over the decision. There content was so crude anyway that the credibility of their conclusions could not survive even a superficial reading (to provide an example, the study on the drainage schemes compared the value of agriculture production before and after drainage, with the hypothesis that all the surface would undergo the maximum possible level of intensification of production… but without factoring any of the costs (investment, subsidies, annual inputs) incurred in the process! (Mermet, 1981). In two other cases – the motorway and the hydropower dam (Henry, 1986) – the economic studies provided in support of the project were quite sophisticated. It required painstaking and skilled analysis to identify how – by which choices of methodological detail and of parameters – the authors of the studies had gone about making them produce the results the promoters needed to justify their projects. For instance, the authors of the economic study on the motorway project used overly optimistic traffic forecasts (in view of the available data at the time, and as facts have since proven) and factored in the benefits monetary values for elements like time saved, and safety gains but also – more controversially and with a major impact on the study's result – a "comfort factor" of driving on a motorway rather than on a four lane non-motorway standard road.

However, whether the economic studies were crude or sophisticated, analyses of the decision-making process showed that they did not play a decisive role. What, then, was decisive? In the case of the industrial waste facility, the successive social, political, legal irreversibilities created by the sequential steps of the decision-making process led to a decision that could hardly be considered rational at all, but rather an example of "absurd decisions" (Morel, 2002) or "planning disasters" (Hall, 1980). But in the three other cases, the driving force behind the decision was clear: a highly structured sector-base policy, aiming at deploying large-scale technical systems (intensive agriculture, the motorway network, and the French hydro-power system), implemented by specialised technical professional bodies, backed by major political forces and supported by ample public funding. Under such circumstances, cost-benefit studies were used as justification, sometimes maybe for intra-sectoral negotiation on investment priority, but as far as interacting with environmental impacts, they were used for advocacy in favour of projects, against environmental concerns and for justification for building them once their construction had already been decided.

A striking result from those case studies, from the point of view of ESV, is that if decision-makers are ready to fund and implement infrastructure projects the cost-benefit balance of which is negative in the order of magnitude of tens or hundreds of millions of euros (or can be manipulated with ease to cover such losses), it is hardly likely that factoring in calculated values for biodiversity could make a meaningful difference.

1.6.2. *But this was France! This was the 1980s!*

Looking at such results today, the objection that may come to mind is: "this was France, these were the 1980s"!

It may be true that in France (and in fact in many EU countries), economic valuation of projects is taken with somewhat more critical distance by policy-makers than they may be in the US for instance (Navrud and Pruckner, 1997). Likewise, within France, they are used more seriously and discussed more in depth in some sectors (energy, transport, although the examples above showed that this is not a general rule) than in others (agriculture, fisheries). These differences can be assigned to a variety of factors such as the attitude of political decision-makers towards economic studies, differing legal frameworks, or the compared influence of engineers and economists in the professional milieux involved. But such differences do not alter the facts: (1) that in all national contexts there are development projects with high negative impacts on ecosystems and controversial economic benefits (Henry, 1986), and (2) that if one looks around the world, there are many political and administrative contexts in which it would be hard to argue that economic calculus is one of the most decisive factors in political decisions.

As for comparing decision-making in the 1980s and decision making today, there may be some stereotypical expectations that economic valuation would play a more decisive role today. But is this really the case?

First, as we mentioned above, we have not found evidence or analysis to that effect in the literature on ESV. The approach that dominates in that literature is a supply-side one, where ESV is promoted by environmental economists as a way to push for better consideration of environmental issues in decision-making. Usually, they plead both for better use of economic valuation in decision-making and for use of ESV as part of this economic valuation. The situation would be very different if the development of ESV was demand-driven, that is, if actual decision-makers would come forward and state that they wish to found their decision on the economic valuation of projects.

Second, recent studies on the use of cost-benefit analysis in decision-making do not support the idea that it would be increasing. For instance an extensive study by the independent evaluation group of the World Bank (World Bank IEG, 2010) shows that the level of use of CBA in the bank's decision procedures has actually been declining since reaching a peak in the 1970s, and that the quality of CBAs used in the Bank's procedures is far from satisfactory. Since the World Bank is a global leader in the promotion of economic rationalisation as a basis for policy-making, this is a very significant finding. Investigating the use and possible uses of ESV in the decision-making of public development banks, – a recent research shows these findings to be confirmed by interviewees in three other development banks as well: the French Development Agency (AFD), the European Bank of Investment (EIB) and the German KfW Bankengruppe (Haddad, 2011).

Based on the evidence available from the environmental economics literature, from the more general literature on environmental decision-making, and from interviews with practitioners, it is reasonable to take as a basis the fact that these issues and difficulties are still pervasive.

All evidence gathered up to now suggests that the implementation gap of ESV is closer to a protracted stalemate, which plays a salient role in our long-standing difficulty in effectively solving biodiversity issues, than to a temporary blockage.

1.7. ESV has no effect by itself

Since Claude Henry's first investigation of the use of valuation in decision-making, we have been regularly engaging the issue of ESV use in decision-making, either as providers of ESV in various decision-making contexts [7], or through research projects on the use of ESV [8]. We have only very rarely come across a situation where the environmental economist, by providing carefully founded valuation of ecosystem services, contributed to dispassionate CBA calculations, or to accounting systems, that would be the main basis for decision-making. A few examples can be cited however. Some jurisdictional cases, such as valuation of damages from the Exxon Valdez oil spill (Carson *et al.,* 2003), can be considered as such: they were commissioned by a court, the demand for valuation came first, and it was at least partially used to back legal arguments and justify juridical decisions. Henry (1990) also describes how valuation was used in a strongly structured participation process in the Netherlands, for deciding over important options in shoreline protection. Another example would be the one that Gowan *et al.* (2006) mention, in a context where ESV was commissioned by supporters of an environmental de-commissioning of a dam in a US river, so as to push the administration to actually implement the legislative decision. In those few examples of effective and demand-driven cases, ESV was however but one element in decision-making processes that had their own various logics. As to the role of ESV, it has been highly variable in these examples, from "one more box to tick" in the routine process of some administrative procedure to providing important arguments in very momentous, contentious and political decision-making processes.

The important point here is that valuing ecosystem services has no impact on decision-making by itself. It has to be relayed into actual decision-making processes. And in that process, the potential role of economists working on ESV is essentially defined by the fact that they are both economists, and specialists of the environment. They are expected (a) to intervene as an environmental voice in the general debate about the economics of decisions that are in preparation and (b) as an economic voice amongst those who speak for environmental concerns.

[7] Mermet, 1990; Cattan *et al*, 1996; Laurans and Dubien, 1996; Bouni *et al.,* 1998; Laurans, 2009; Laurans and Aoubid, 2010.

[8] Marmet and Grdnjean, 1983, Laurans, 2000; Godard and Laurans, 2004, Laurans *et al.,* 2013.

Conclusion

To sum up, ESV has been developing over more than five decades, and has recently experienced a sharp increase in both the effort and the attention it gets. The community involved in ESV, however, is gradually coming to grips with the implementation gap that separates the high potential assigned to ESV and disappointing levels of actual use and impact (and particularly of well-documented cases). As it does so, understanding the decision-making processes contexts and processes that ESV must impact to make a difference in favour of biodiversity becomes a priority. This will be the main focus of the book from chapter 6 onward. Once ES valuators get a better sense of the wider games of which ESV is but a part, two avenues are open for improvement: more rigorous valuations, or valuation processes that are more open to societal voices and concerns, and more adapted to the intricacies of decision-making processes. Although the tensions between promoters of each of these two paths may be acute at times, there is not really a dichotomous choice to be made here.

Evolutions of the methodologies in the *Policy evaluation* domain, which is a related field of research and practice, is a useful parallel, to illuminate what we see as the next steps for the ESV field. Quantitative approaches to policy evaluation, which claimed to objectively measure the effects and efficiency of policies, dominated through the 1970s. Then a very strong movement developed in the 1980s to design and promote approaches based more on the evaluation process than on objective content only (Guba and Lincoln, 1981). The polarity between these perspectives still plays a structuring role in the field. But since the 1990s, it has gradually matured into a palette of approaches that combine in different ways attention to measurement and attention to process. A call to focus on utilisation (Patton, 1986) has played an important role in that transition: policy evaluation, just as ESV, is justified by its ability to *make a difference* – for which actual use is a necessary condition. Extreme posturing on one side or the other has subsided and the focus of interest is now rather in the fit of measurement methods to process and needs, and of the overall framing and approach of each evaluation to specific context. The variety of approaches that is apparent to those who are familiar with that field testifies to the fact that as one renounces the "one size fits all" approach, and acknowledges that process may be relevant, the need to adjust to contexts triggers more and more precise and diverse methodologies.

Our review of the literature and discussions with practitioners and experts seem to indicate that a similar transition is perhaps beginning in the ESV field. The focus on use and contexts of use may both facilitate the transition and start building essential resources for delivering the promises of ESV for better taking biodiversity into account.

ESV is however an ETB (Economic tools for biodiversity) that originates in the academic field. This may partly explain why its demand-driven uses seem so rare. The next chapter will turn to a very different kind of ETB, Payment for Ecosystem Services, which is conversely a well-developed practice on the field, and has received only recently enthusiastic attention in the environmental economics literature.

2. Paying for ecosystem services: a simple concept, a complex practice

Whereas valuation of ecosystem services remains a somewhat abstract exercise in the economics of biodiversity, economic instruments become much more concrete when they involve "putting money on the table", as do payments for ecosystem services (PES) which we will examine in this chapter. The principle of PES is quite simple: "*governments and private entities can pay resource owners/users to protect natural ecosystems or adapt management practices to enhance provision of ecosystem services.*" (ten Brinck, 2011, p.181).

Neither the principle nor the practice of such payments are a recent invention. They have existed for decades for instance in the field of water/watershed management (Westman, 1977; Barraqué, 1992). What is new is the remarkable salience PES have taken on in the contemporary debate on how to solve biodiversity issues. Several developments have converged over the last decade to give PES their current central position in the range of solutions to be considered. First, in the wake of the promotion of the "Ecosystem Services" concept by the MEA and of publications advocating payment for ES as a powerful lever in favour of biodiversity (Daily, 1997), "PES" has become a federating label bringing together quite heterogeneous mechanisms, giving them increased visibility, at the risk of some confusion. Second, the increasing attention given to economic tools, and the parallel promotion of private involvement in environmental policies, combined with rising critiques on administrative action, have pushed PES higher on the agenda. A third factor is probably the fact that over the last decades, PES systems, which had developed quietly in developed countries, have been promoted vocally for countries of the South. Discussions on mechanisms such as REDD+, which is to fund ecosystem services rendered by forests by storing carbon, but that may have large-scale effects on the funding of biodiversity, have attracted much attention from both stakeholders and academics (Scheliha *et al.,* 2011). On smaller scales, PES in the South now also appears as an attractive source of funding at local or regional scale, alternatively or as a complement to more traditional development aid funding, in contexts where sources of funds are often quite limited (Pearce, 2007). In brief, the prospect of new financial opportunities, and the hope that one set of economic tools would break beyond the limits of current environmental actions, have attracted a massive amount of attention on PES.

As PES have risen on the environmental policy agenda, their presence in environmental and environmental/ecological economics literature has soared. In just two years, Ecological Economics has devoted no less than three special sections to PES, as Joshua Farley and Robert Costanza (2010) point out in their introduction to the latest of these series of papers. As we examine this expanding literature from the point of view of the use of ETBs, and of the connection between theory and practice, three features are quite apparent: (1) there is an abundant body of examples of payments for ecosystem friendly practices. Somewhat in contrast to what we described in the previous chapter about ESV, where documented use for decision-making is sparse, authors on PES have been actively looking for implementation cases from the ground, and they have found many concrete examples – when they are not themselves part of the projects they analyse (Landell-Mills and Porras, 2002. ; Wunder *et al.*, 2008): this is indeed a field with extensive engagement of researchers on the ground. Here, practice, as it were, precedes theory. (2) as a consequence, and again in contrast with ESV, analysis of PES schemes on the ground and debate about what they can contribute in general to biodiversity is often quite sensitive to the contexts in which the tool is used. Conservation experts, economists and stakeholders who are considering paying for ecosystem services all converge towards two linked concerns: is providing a payment really going to solve the biodiversity challenge and is it going to make enough difference to justify the expense? All (or almost all) are aware that the answers are very context-dependant. (3) Much attention is given in the literature to the issue of defining and classifying PES and it is apparent that authors are truly struggling with that issue. As Wunder *et al.,* (2008, p. 839) write in their introduction to a large review of PES experiences: "i*t becomes a judgement call as to whether several individual programs should be considered "PES with qualifications", or "non-PES with PES-like characteristics*". [...] *Even among us three editors, there is thus some disagreement over where exactly the line between PES and non-PES should be drawn.*" This perplexity can derive from the huge diversity of arrangements that imply some form of payment for some sort of ecosystem-friendly use. Besides, much of the literature on PES tends to define it in a rather narrow way, which contradicts the huge diversity of operative mechanisms that are currently grouped under the PES label.

Examining this tension between the narrow, "archetypal" definition of PES and the variety and complexity of PES implementation situations on the ground will provide the common theme of this chapter. Using and following results of a one year research project that we led on PES in 2010 (Laurans *et al.*, 2012), we will start by providing examples of PES schemes. We will then turn to the archetypal definition of PES and examine why it is so influential in the field even though it struggles so much to encompass the variety and complexity of PES practice on the ground. The bulk of the chapter will then be devoted to linking up principle and practice successively on each of four major components of the PES principle: Who is the buyer? How direct is the transaction? Who is the seller? And what does the payment buy? Finally, expanding on findings, we will discuss issues that arise from debates on the current and envisaged practice of PES, in particular the relation between PES and other conservation tools used in conjunction with them. We will discuss in particular the risk that payments be taken opportunistically without leading to the expected conservation results, the pitfalls involved in the current plans for massively scaling up the implementation of PES and the risks involved for the balance between various concerns of environmental policy (such as climate change mitigation and biodiversity conservation, for instance).

2.1. Examples of PES

Since PES practice has tended to precede theory and remains, in our view, rather more complex than theory, it seems appropriate to start our discussion by some examples from the ground.

2.1.1. *Paying the Maasai for wildlife friendly management of their land in Eastern Africa* [9]

Our first example is located in Eastern Africa in a Maasai village adjacent to a National Park. The habitat required by the wildlife in the park extends much beyond the park's limit: large herds of ungulates – followed by their predators – use migration routes and foddering ground outside the park, in the land of Maasai cattle-breeders. This wildlife has been declining over the recent years, due to both the penetration of farming in some areas, instead of traditional extensive grazing, and to poaching of wildlife for commercial purposes. The PES project was launched by a tour operator, an hotelier, a conservation consultant and an elephant researcher. They all shared

[9] This example is taken from a case study analysed by Ecowhat (Yann Laurans) for AFD (Laurans *et al*, 2012), http://www.afd.fr/webdav/site/afd/shared/PUBLICATIONS/RECHERCHE/Scientifiques/A-savoir/07-VA-A-Savoir.pdf).

the same diagnostic: (1) governmental policies for wildlife environmental protection were inadequate; poaching activities, in particular, were hardly curtailed; (2) nature reserves, despite being well protected, were unable to ensure the proper functioning of ecosystems and more specifically did not allow for the zebra and wildebeest migrations that contribute to the quality of the ecosystem; (3) the cultivation of lands bordering the park tended to increase due to farmers coming from other regions who were gradually replacing the Maasai cattle-breeders.

As a result, the group, acting through a local NGO serving as an intermediary, offered a simple "business deal" to a nearby village council: a yearly payment of around USD 3, 000, on condition that the village protected its lands from any attempts to convert it for farming. A five-year contract was signed with the village in 2008, which remained the sole owner of the land (the land is collective property in the country and cattle breeders do not hold individual ownership rights over grazing areas). The village accepted the agreement (it was signed by all the village council members). The first yearly payment was used by the village council to go to court and retrieve their property rights over land that was being illegally farmed by a foreigner who had appropriated the land.

This example is a private initiative, to the point that it was undertaken without the knowledge of local administrative authorities because the parties are apprehensive about misuse of funds and corruption if the latter were officially informed and became involved in the process, and because such private deals are prohibited by the national authorities. It was worked out as an ordinary business deal. All the financing parties involved claim that they were motivated purely by their own economic interests and not any altruistic reasons: in this case, biodiversity attracts the highest-end of international tourism and for some localities, represents a major, if not essential, source of livelihood.

However, it is to be noted that other actors were also extensively involved: an elephant researcher, a NGO consultant, a NGO promoting local governance, etc. Indeed an important factor in the villagers' readily accepting the agreement was the NGOs offer of a second agreement providing for the employment of four anti-poaching village scouts (at the NGO's expense, mainly funded by donations from the United States). Moreover, the system worked because the court judgment over the encroaching farm was awarded in their favour, and there was hardly any expenditure and no corruption in the jurisdictional process.

Whereas this case is – and intends to stay – very discreet, our two next examples have been widely publicised and used to promote the use of PES.

2.1.2. *Ecosystem services as precious economic capital: The French Vittel success story*

Just as the above example from Maasailand, the Vittel case is very local, but in a completely different context.

Though based on just one spring in north-eastern France, at the foot of the Vosges Mountains, Vittel's bottled water is sold internationally and supports a large industrial bottling operation[10]. By the end of the 1980s, the mineral water produced by Vittel showed an increasing content in nitrate and pesticide levels. The cause was easily identified as intensification of agriculture in the spring's watershed. In 1989, Vittel funded a large action-research project on its catchment led by the National French Agronomic Research Institute (INRA, "Institut National de la Recherche Agronomique").

The aims of the project were (1) to ascertain and map with precision the role of various farming practices on nitrate leakage from farmland in the catchment and how that leakage impacted bottled water quality and (2) to investigate, in collaboration with farmers, what changes in practice would limit leakage to target levels, how they would fit in farm management and economy, and what the opportunity costs would be. Based on that research, INRA was able to recommend a set of practices that would allow reaching the targets in terms of water quality. The practices consisted in: complete elimination of corn crops, composting of all animal waste, limiting livestock density to 1 unit[11]/per ha of the cattle grazing area, prohibition of utilisation of plant protection products, limiting nitrogen fertilisation to composted manure in amounts that plants can really use, introducing a new alfalfa-based crop rotation, modernisation of farm and cattle-breeding buildings so as to limit leakage, facilitate composting, etc.

Based on these recommendations, Vittel launched an agricultural conversion program in the 1990s, implemented through a subsidiary, Agrivair, created to that effect. The program combines: (a) long-term (8 to 30 years) contracts with the farmers on farming practice, (b) yearly payments (228€/ha) to farmers for a 7-year period as they adopted the new practices, (c) free services to farmers, consisting both in technical assistance and in free spreading of manure, (d) a one-off payment for the initial conversion of the farm, to cover for changes in buildings or equipment, (e) a land-purchase program, whereby Vittel gradually purchased most of the land and then, on condition that they accepted the contracts, offered it back to the farmers with free leases.

[10] Vittel was an independant company until 1992, when it was bought by Nestlé and became a part of its Nestlé waters division.

[11] Unit = equivalent of 1 head of cattle.

The program was estimated to have cost Vittel the equivalent of EUR 0.15 per litre of mineral water, against an expense of about EUR 24.25 million (Perrot- Maître, 2006; Deffontaines et Brossier, 1997).

After these two local examples, let us turn to a third and final one, on a much larger scale.

2.1.3. *Costa Rica, a hallmark example of a large scale PES system* [12]

The Costa Rican national Forest Law mentions four environmental services (ES) provided by forest ecosystems that must be exploited in a sustainable manner: climate change mitigation, biodiversity conservation, protection of catchment areas and conservation of landscapes. Since 1997, the "Pagos por Servicios Ambientales" program has been making compensatory payments to more than 4,400 farmers and forest owners to improve reforestation, sustainable management and forest protection. The payments take the form of multi-annual contracts (often over 20 years). New tree plantations, the development of related activities, sustainable felling, etc. are remunerated. A national Forest Fund ("Fondo Nacional de Financiamiento Forestal", FONAFIFO) was created to support this mechanism. Its funding comes from a tax on fossil fuels sales, incomes from hydroelectricity sales, World Bank loans and Global Environment Facility (GEF) grants (Pagiola, 2005; Steed, 2007). This case was widely publicised by the World Bank, (Stefano Pagiola, in particular) as being an innovative and successful example of biodiversity management. Although the State was at the helm of the mechanism, it is to be noted that the funding was not provided through the country's budget but by means of an earmarked tax, specially created for this purpose. Moreover, the funds were intended to pay forest landowners directly so that they adopt sustainable forest practices.

2.2. How much does the attraction of PES depend on their definition?

The three examples of PES we just presented illustrate the important differences in scales, contexts and arrangements across the many heterogeneous cases that are currently grouped under the PES label. This diversity is central to the discussion of the use of PES and will also be guiding our discussion here.

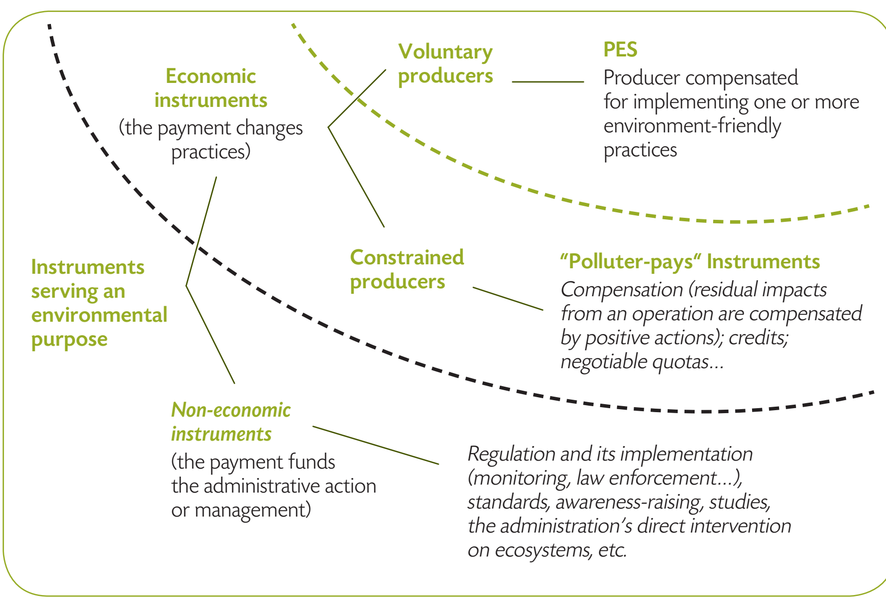
[12] This case is among the most commented PES mechanisms. See for instance: Grieg *et al., 2005.*

2.2.1. *An archetypal definition of payments for ecosystems services*

As an introduction, we shall start from Wunder's (2005) often cited definition of PES as (a) a *voluntary* transaction where (b) a *well-defined* environmental service or a land use likely to secure that service (c) is being "bought" by a (minimum one) service *buyer* (d) from a (minimum one) service *provider* (e) if and only if the service provider secures service provision (*conditionality*).[13]"

Although the features of this definition are central in current discussions of the potential of PES for conservation, it is striking that, of the large number of cases documented and discussed as PES, hardly any really conform to this definition. The only common point between these cases seems to be that ES sellers are always voluntary (they are not constrained to enter the system) and that the system implemented involves an economic instrument through a payment (Laurans *et al.*, 2012) (cf. Figure 2 below).

Figure 2 *Differentiating PES from other instruments serving environmental purpose*



Source: adapted from Laurans et al., *2012.*

[13] The emphases are the author's.

In the same publication (p 4), Wunder himself notes that "*While the number of tropical PES-like initiatives is thus considerable – Landell-Mills and Porras 2002 reviewed 287 such schemes – there are probably very few "true PES" conforming to the theoretical concept developed in the literature and described in the simple definition above.*"

This tension between the (heterogeneous and complex) practice of paying for ecosystem services and its (precise and restrictive) definition and labelling is at the centre of the PES debate. Interestingly, this is not a contradiction that is pointed up by critiques of PES, but one that is widely recognised and freely discussed by authors who are promoting them. Still in the same publication where he proposes the oft-quoted definition, Wunder writes: "*if nevertheless, I prefer to maintain the above pure definition, it is out of a belief that these five principles represent something new – a more direct approach that deserves to be tested on its own terms*" (p4).

In other words, there is more than a definition at stake in the archetypal concept of PES that is the central reference in the current debate, there is a promise. And the tension between the practice and theory of PES is one between that promise and the possibility of delivering it on the ground. Let us now examine that promise engrained in PES, before putting it in the context of real-world action and strategy.

2.2.2. *PES: a promise to move beyond the disappointment with conservation policies*

PES are often presented as way to break away from other conservation tools that are pictured as having failed and still failing to curb biodiversity loss. A very obvious target of such criticism are "command and control" policies and tools that would enforce conservation through regulatory means (Damania and Hatch, 2005). But also under fire are strategies based on "Integrated Development and Conservation Projects" and "sustainable ecosystem management projects" that, since the 1990s, have tried to incorporate social and economic objectives into their ecological vision (Margoulis and Salafsky, 1997; Brandon *et al.*, 1998; Hughes and Flintan, 2001).

By the end of the 1990s, many started indeed to feel that, like traditional policies, these projects were also unable to prevent biodiversity erosion by themselves (Balmford *et al.*, 2003). First of all, PES promoters saw both these IDCP projects and the traditional policies as unable to bring about lasting changes in land use, however action over land use appears as an essential condition for biodiversity conservation (Rice *et al.*, 1977; Brandon *et al.*, 1998). Secondly, they claimed that public financing for biodiversity conservation has remained woefully inadequate to achieve such an objective (Pearce, 2007). Thirdly, they saw regulatory and administrative policies as far removed from the local issues and from local users (Damania and Hatch, 2005). Finally, they underlined that projects and policies led to a build-up of taxes and standards that were difficult to enforce (Economic Commission for Europe, 2006).

These shortcomings of public policies aimed at biodiversity conservation therefore gave rise to considerable criticism, bringing four basic requirements to the fore, designed to address them:

1. the need to implement measures that would impact on land use.
2. the need to mobilise new funding sources to compensate for low public budgets.
3. the need to implement instruments at a local level and on a pragmatic basis.
4. to do all this without creating new regulatory instruments.

PES systems appeared then as a viable option to counter these shortcomings (Redford and Adams, 2009).

Indeed, the principles embedded in the archetypal definition of PES are promises to answer those challenges:

1. the insistence on a well-defined environmental service and verifiable conditionality (points (b) and (e) in Wunder's definition) answers concern over a lack of tangible impact (Pagiola, 2008).
2. the mobilisation of new "buyers" (point (b) in the definition) carries hope of solving issues of inadequate funding sources (Pearce, 2007).
3. paying the "sellers" as directly as possible (as implied by the verb "bought" in point (c)) has potential for more local, pragmatic and less costly solutions – paying someone for a service is more straightforward than voting and enforcing laws, or than launching complex integrated projects (Kemkes *et al.*, 2009).
4. the voluntary character of PES (point (a)) bars the perspective of still more regulation (Economic Commission for Europe, 2006).

Such then are the principles giving PES their potential to operate a breakthrough in biodiversity conservation, and to be seen as "*arguably* [...] *the most promising innovation in conservation since Rio 1992*" (Wunder, 2005, p. 3).

It is worth noting that in their archetypal form PES are attractive to very different actors in the field of biodiversity policy. For economists, the link between payment and results, as well as directly negotiated payments, make PES fall in their purview and fulfil conditions for treatment of problems in the economic (rather than administrative) realm, with the associated promises of efficiency. For conservationists, the pragmatic aspect involved can be felt as opening new avenues for action in a context of perceived impasse, avenues of action where they have important roles to play. For stakeholders of rural development, PES attract attention as a new source of funding, especially for areas that are left on the roadside of mainstream development efforts and funding. For everyone else, PES promise minimal disturbance as it does not compete with existing funding nor interfere with existing legislation and governance structures.

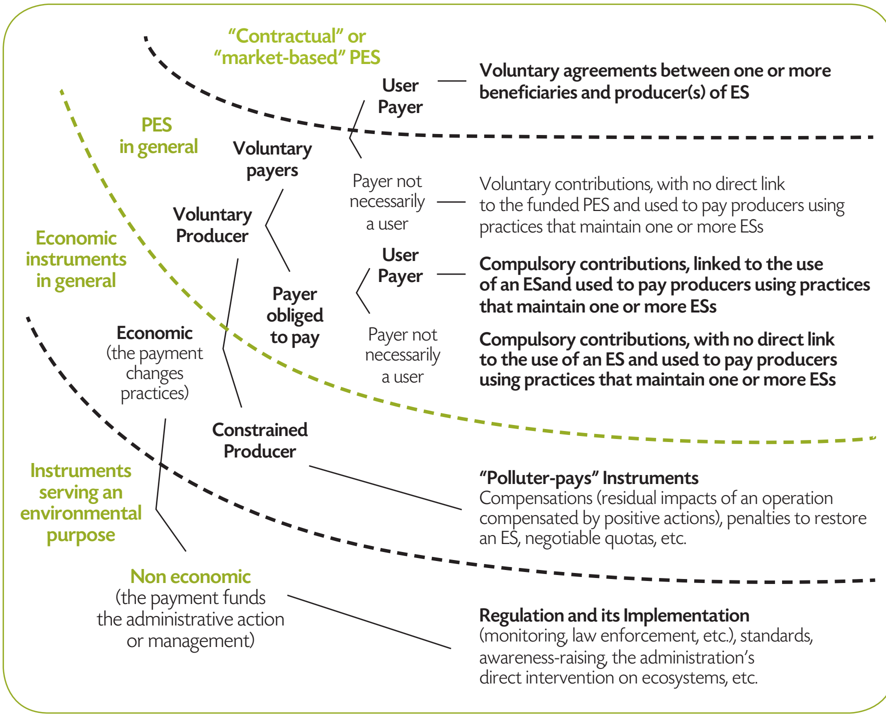
In view of this convergence of hopes, one is less surprised by the flurry of attention PES have received over the last decade, or by the willingness of many to live with the tensions between the archetypal model and realities on the ground, or by their efforts to try to work these tensions out. Let us then examine in more detail, based on the literature and on our own case studies, what these tensions are, and how they are managed, by questioning in turn each of the four main principles of the archetypal definition of PES: Who is the "buyer"? How direct is the transaction? Who is the seller? And what exactly does the payment buy?

2.3. User or not, voluntary or not: who is the buyer makes crucial practical differences

The first contribution expected from PES is to bring into the biodiversity management scene new buyers who are prepared to pay in order to conserve or restore ecosystem services. Of course, new funds are the central aspect of their contribution, but not the only one. By choosing which services they are ready to pay for (and which they aren't), by choosing sellers and negotiating with them, they may also become an organising force in biodiversity conservation and restoration. But who are these buyers?

Examining if the buyer is an ES user or not, and if his payment is voluntary or not, led us to differentiate four categories of buyers and therefore four categories of PES (Cf. Figure 3).

Figure 3 *A four PES categories Typlogy*



Source: adapted from Laurans et al., *2012.*

2.3.1. *Buyers are ecosystem services (ES) users and their payments are voluntary*

In the archetypal PES, they are ecosystem services users who are ready to make *voluntary* payments directly to "sellers". Our example from Eastern Africa illustrates this situation: a few local eco-tourism operators who need and use the wildlife in their activities, strike a "business deal" with a Maasai community. They do so of their own initiative, without any intervention from government. The Vittel case is quite similar. That case is that of but one buyer for whom the protection of water quality is a service of direct and high economic value. This is also the case for instance in bio-prospecting agreements, in which pharmaceutical and biotechnology companies decide to pay suppliers who control biodiversity-rich ecosystems to obtain exclusive property rights over useful components that they might extract from these ecosystem. In examples such as the much advertised Costa Rican system for bio-prospecting, the buyers are private economic agents (cosmetic or pharmaceutical firms), the system is well organised and clearly determined services are marketed by the selling institution, the National Biodiversity Institute.

2.3.2. *Buyers are not specifically ES users and their payments are voluntary*

In many other cases, however, the buyers are not any more specifically users of an ecosystem service but their payments are still *voluntary*. To illustrate such cases, let us take the example of a bird nest protection scheme in Cambodia, as documented by Clements *et al.* (2009).

The northern plains of Cambodia are home to numerous bird species, which are increasingly endangered as the local people collect eggs and chicks for trading purposes. A bird nest protection program aimed at saving birds from extinction was therefore initiated in 2002, in four villages of the Kulen Promtep Wildlife Sanctuary (Clements *et al.*, *ibid*). It is conducted by the Wildlife Conservation Society, WCS, an environmental NGO, which raises funds through donations.

Under the program, rewards of up to USD 5 are offered to local people for reporting a nest. The villagers can also monitor and protect these nests until the fledglings are ready to leave it. The program encourages them to refrain from harmful practices such as egg consumption, cutting down of nesting trees, etc. They receive USD 1 per day for this work. If the service provided is successfully achieved (*i.e.* the chicks leave the nest), the payment is doubled. The payment amount was determined after consulting with the villagers. Contracts are signed individually. WCS monitoring staff conduct weekly monitoring visits. The total cost of the program for WCS is around USD 25,000 per year and the average cost per nest protected ranges from USD 65 to USD 120. The average cost has declined as the number of monitored nests has increased and monitoring costs can be shared by adjacent sites. 71 to 78% of the expenditure has been directly allocated to the local population, while monitoring costs account for the rest. The average payment per family is USD 100 per year. Some villagers have specialised in protecting the nests, switching species depending on the nesting season. This enables them to obtain aid through most of the year, with incomes of up to USD 400 per year, a significant source of income as compared to the other income sources available.

Between 2002 and 2008, more than 1,200 nests of threatened species have been protected. Many species' populations have actually increased. However, for some species, the numbers have remained constant. This indicates other major threats apart from hunting: habitat destruction, the direct effect of deforestation and the intensification of agricultural practices.

In this example, there is not much sense in seeing the NGO and its members, who fund the PES, as ES "users". They are however willing to contribute to bird conservation. Here, the originality of PES as a tool is not in the fund-raising (from ES "users"), which is done in a classic way, but in the delivery of conservation action, which is done by directly remunerating practices that are good for conservation.

2.3.3. *Buyers are ES users and their payments are not voluntary*

Moving one more step away from the archetypal PES, we move into cases where the "buyers" are, as in the first above case, ES users, but are not at the initiative of the deal : their payment is *not voluntary*. Here is the example of a water management scheme on Lombok Island, Indonesia [14]:

This case study concerns Lombok Island, which has a surface area of 5,435 km2, and forms part of the Indonesian province of Nusa Tenggara Barat, to the east of the Indonesian archipelago. The forests located on the Rinjani volcano and nearby play a crucial role in the hydric regulation of land in the island's north and in limiting erosion on the volcano's slopes.

Due to increased land clearing and cultivation, extensive deterioration of water flows, along with forest deterioration, have been observed. Between 1992 and 2002, 43% of the springs around the volcano dried up and nearly 30% of the woodlands on the volcano's slopes have disappeared.

From 2004 to 2007, under the aegis of the United States Agency for International Development (USAID), negotiations aimed at enforcing corrective measures took place. The formula that was initially adopted was based on voluntary fund-raising but it was decided in the end to impose a compulsory levy upon all water users.

The agreement created a multi-stakeholder agency, called the Multi Party Institution (MPI) which is responsible for managing the financial resources and entering into agreements with the producers using the volcano's land. Most of the stakeholders, in particular the residents of the downstream city, insisted upon the creation of the

[14] This study, conducted and written by Romain Pirard, Institute for Sustainable Development and International Relations (IDDRI, Institut du Développement Durable et des RelationsInternationales), is partly based on Pirard (2012).

agency because they did not trust the State or the territorial authorities to manage their funds in view of the high level of corruption and the State's poor record in managing public money.

Fund-raising started at the end of 2009 and the amounts raised so far correspond remarkably to the estimations that the MPI had anticipated. However, things did not move as fast when it came to payments for landowners and upstream farmers: for several years, thanks to the financial support of developmental agencies and/or the Indonesian government acting through various programs, pilot activities were undertaken to restore the degraded lands but despite new funding, no PES contract has been finalised as yet. This is mainly attributed to the lack of capacities and of human resources assigned to the contract negotiation and formalisation process. Hence, the situation is not due to any lack of motivation or will on the part of the service providers – in this case, the rural population.

It is interesting to note in passing that this is a rare case of PES where monetary valuation of ecosystem services was used and seems to have affected the outcome. In 2001, a valuation study, funded by USAID, was instrumental in prompting the establishment of the working groups of stakeholders which eventually led to the creation of the PES mechanism. However, in line with the findings in the previous chapter, the ESV was used as a resource for advocacy in favour of collective action, not to technically adjust payment levels.

This example from the South appears as an innovation. But systems based on non-voluntary payment by water users for water-related ecosystem services protection have been wide-spread in the North for a long time. A well-known example is the Catskills of New York State (US) (Hoffman, 2010), an emblematic example repeatedly cited to illustrate the potential of PES, and to which we will return later.

2.3.4. *Buyers are not ES users and their payments are not voluntary*

A final step away from the archetype will lead us to situations where the buyers are *not users of an ecosystem service*, and their payment is not voluntary. Here, a heterogeneous set of fee- or tax-payers provide the money for buying ecosystem-friendly practices. The famous example of agro-environmental payments of the European Common Agricultural Policy falls into this category, since the payers are the European taxpayers in general, who are not necessarily users of the biodiversity services bought by the subsidies.

2.3.5. A wide array of buyers paying for various ecosystem services

Overall, from looking at many examples either through the literature or directly on the ground, there appears to be a large variety of situations with regards to who the buyer can be.

We also have to note that PES systems can and usually do combine several of these buyer categories as shown in the following example of the Saltillo Basin in Mexico [15].

The NGO Profauna works for the preservation of the Saltillo watershed in northern Mexico and more particularly for the conservation of Zapalinamé Forest, which contributes to the restoration of part of the basin's water resources. To do this and other tasks, it pays local voluntary forest owner communities for them to implement forest-friendly management practices which favor the watershed's hydrological functioning and the conservation of the biodiversity. So-called "PES" contracts are drawn up between the NGO and these communities for periods of 1 to 15 years. The funds used by the NGO to pay for these contracts come from diverse sources, thereby combining in the end several of the various kinds of buyers that have been reviewed so far:

- Part of the money comes from voluntary donations by users of Saltillo's water (first category reviewed above);
- Another part comes from donations from the Mexican Nature Conservation Fund (FMCN), itself funded by the Gonzalo Rio Arronte Foundation (FGRA) for this project (second category reviewed above);
- A third part comes from a mandatory tax imposed by the Mexican government on drinking water distribution companies (third category reviewed above);
- Finally, part of it comes from subsidies from the Saltillo municipal corporation, and another part from Mexican government subsidies (fourth category reviewed above).

As we saw, only in a limited number of cases does the buyer conform to the archetypal PES model of a user paying on a voluntary basis. However, as long as the considered payment schemes do mobilise new ES motivated funding, the situation is worth considering in the light of the PES model, as it delivers some of its promises. But this

[15] This example is taken from a case study analysed by Tiphaine Leménager (AFD) ((See Laurans *et al.*, 2012).

does not mean that any environmental action involving payments should be considered a PES. As we already mentioned earlier and as Figure 2 posits, there is little point in including in PES and PES-like instruments, tools which producers are constrained to provide, or schemes where economic incitation (the payment motivating change) is not the active principle for change. This is a point agreed by the many authors working on PES.

After examining the variety of PES – or PES-like – situations in terms of who the "buyer" is, lets us now turn to another component of the "promise" of PES: breaking free of the ponderous processes of command and control and polluter-payer policies, as well as of multi-faceted, multi-stakeholders integrated development projects.

2.4. How direct is the deal? Intermediaries shape PES

Some of the main promises of PES systems rely partly on the directness of the transaction these systems are expected to generate. Also called a "short loop" effect, this directness should indeed allow local and pragmatic solutions that are quicker to implement and less costly than any command and control policies or integrated project. But what about this short loop effect in practice?

2.4.1. *The attraction of the short loop*

Examples of PES cases with very short "loops" are often used to support the feasibility and the attraction of such directness.

Let us consider for instance the Vittel case. The Vittel scheme constitutes an exception in a French context where public action to curb water pollution and damage to aquatic ecosystems from agriculture engulfs significant financial and administrative resources with limited success (Saleth and Dinar, 2000; Biswas, 2004; Jeffrey and Gearey, 2006). Amongst the many reasons for this lack of success of public policy, the resistance of farmers' organisations and of the agriculture ministry plays a very important role. In the Vittel success story, directness lies not only in direct contact between the water bottling company and the farmers but also, maybe more importantly, in not going through the channels (and the political and other hurdles) of public policy and the farmer's union.

2.4.2. *However short the loop, intermediaries are crucial*

This does not mean, however, that the deal was direct in the sense that it would do without intermediaries. In the Vittel case, it took skilled intermediation to reach the deals that satisfied the different actors' needs. The action-research project contracted to INRA played several roles in this process. First, the project started by sociological investigation to help the company understand the farmers and view the situation from their perspective, not only in technical and economic terms, but first in human, sociological, personal, local, historical terms. Second, the project then helped start roundtable discussions between the company and the farmers which led to agreement on the principle of actively researching a direct form of solution. Third, the project was instrumental in understanding the science of the ecosystem functioning and services involved and establishing clear connections between what the "sellers" would do and what the "buyers" would get in a PES deal. Fourth, the project worked out, in the context of a gradually established collaboration with the farmers, the technical, organisational and economic options for delivering the services Vittel crucially needed and was ready to pay for. After all these contributions from the project, Vittel created a subsidiary and hired a member of the research project as director. Serving as an intermediary between Vittel and the farmers, the subsidiary processed the contracts, the payments, the technical advice and interventions. It played a crucial role in the day to day implementation of the PES deal.

This case illustrates that intermediation is key in PES deals. The case encompasses in a nutshell the various sorts of intermediaries that seem to be needed to make a PES scheme work: cultural mediators, scientific expertise on the ecosystem services at stake, technical expertise on the means for their provision, negotiating and legal processing for contracts, financial processing of payments, monitoring and evaluation of performance. At the scale of the Vittel area (approximately 25 moderate-sized farms), this is indeed a very strong presence of intermediaries.

Even the most direct deal we have seen – the Maasailand example above – relies on intermediaries: the elephant scientist, the consultant and others connected the actors, the US NGO that provided additional funds, and the local NGO that buttressed the deal by adding the hiring of guards from the community for surveillance of poaching.

Overall, the directness of PES is neither an absence of, nor a simplified role for, intermediaries and social-political processes. Rather, the promise of the PES here lies in using players and processes differently from those of other instruments. In this sense, PES open up a space for new chains of intermediaries, that may deliver results in some cases where other instruments using other chains of intermediaries would not have done so.

2.4.3. *The loop is not always short*

Another point made by the Vittel case and partly linked to the fact that intermediaries are crucial, is that decision making is usually not that simple and rapid. The automatic alignment of behaviours following a rationally established financial incentive may look impressive in textbooks, but it hardly reflects the actual processes of establishing PES deals. And such a simplification would be a poor guide to action on the ground.

Directness should be taken to consist in the capacity to make decision-making loops appropriate to an ecosystem management situation and no longer than required by that situation. Often, as in the Vittel case, the contribution of PES lies in different and additional loops of action rather than in particularly short loops. Many of the PES schemes we have examined involve decision-making processes that are far from short. Let us consider a second example about a rice certification program in Cambodia proposed by Clements *et al.* (2009).

A community-based, agro-environmental payment program was started in 2007 in Cambodia in order to safeguard the highly endangered local biodiversity. Under this program, rice growers, acting voluntarily, agreed to adopt environmentally more friendly production practices and to stop hunting. They were then allowed to sell their produce certified Wildlife-Friendly [16] through a village committee to a marketing association specially created for this purpose. The latter sells its rice directly on the local open market as well as to tourist hotels which ensures the farmers obtain preferential prices. The program is implemented through oral or written contracts between the farmers and the committee which acts as an intermediary.

To counteract this program and win back the farmers, “traditional” intermediaries decided to increase their purchase price for rice. But most farmers continued to sell their products through the certified market. They indeed revealed that they preferred selling “to their own people” rather than to outside intermediaries as there was greater transparency in their dealings and they had “control over their own future”.

[16] The certification is an assurance that the consumption of the product (in this case, rice) promotes nature and is wildlife friendly.

In this example, the key is the organisation of an alternative marketing channel for rice. It is not really shorter than the one in place before, but is indeed different and allows stakeholders who are crucial for good ecosystem management to exert more control. This example is also useful because it is a small-scale case of an environmental certification mechanism. Better known certification schemes, like the FSC (Forest Stewardship Council), operate on a much larger scale, involving "buyers" and "sellers" on scales that go from national to global. Certification schemes are arguably a form of PES and partake of their essential operative traits. They bring in new "buyers" as buyers of wood products, for instance, pay a premium that is intended to remunerate special care taken of ecosystems and their services, through certified forestry practice. They also introduce alternative decision-making processes and management systems that come in addition to other forms of forest or biodiversity policy (Cashore, 2002; Guéneau, 2011). The loop is long here between the "buyer" (the consumer) and the "seller" (the farmer), but there is still an element of directness, if that is taken to mean short-cutting some of the hurdles of other channels. The issue is not to deal without channels, but to create new ones.

Other examples of PES that do not rely on certification also illustrate long loops and complex management systems, such as the Costa-Rican payments for ecosystem services from forests we briefly presented at the beginning of this chapter.

To sum up, directness in PES deals is important but it does not lie in simplicity and rapidity. This fact gives little ground for arguing, as many do from a theoretical point of view, that there would be less efforts and costs in intermediaries here than there would be in implementing command and control policies or an integrated environment and development project on a similar scale (Engel *et al.,* 2008). Making PES deals takes time. It is fraught with pitfalls and requires expertise and resources that have to be up to the challenge of each case with its scale and specific issues.

One factor that may have played in favour of the surge of interest for PES may be that many persons and organisations have recently acquired the motivation, the capacity and some means to contribute to solving ecosystems and biodiversity issues. However, they cannot all (or they do not all want to) be part of the administrative, judicial, institutional system that implements and controls previously existing policies. Given this context, we can believe that new channels of action such as PES, which are rather independent from those systems, have offered interesting perspectives to those actors.

The operative aspect here – and, we believe, in many successful schemes involving PES in general – may not be so much the shortness of the loop. It might rather be its capacity to by-pass some hurdles of other tools and management systems and activate renewed organisation for decision-making and action. Such configurations have indeed opened (at least in these successful cases) opportunities for good ecosystem management that were not available through other tools and arrangements.

When analysing PES cases, it is thus important (a) to measure the strategic sensitivity, complexity and specificity of the situation, (b) to examine the various necessary intermediary functions that have to be fulfilled, (c) to see to what extent they are fulfilled now and (d) to examine how they can be complemented or by-passed if necessary, how, by whom, at what costs and with what conditions for lasting operation.

2.5. Biodiversity can't always be sold as straightforward goods and services

To continue with our examination of the components of the archetypal definition of PES and of the way they relate with the operative traits of PES on the ground, let us now turn to the "seller" of ecosystem services and start with the example of the Bush Tender program in Australia.

The Department of Natural Resources and Environment (DNRE) of the State of Victoria in Australia has initiated a pilot program called "BushTender", aimed at improving the management of native vegetation on private land. In return for State payments, landholders agree to fencing and managing native vegetation for a period of three years. The first contract under this program was signed in 2002 in the northeastern part of the state of Victoria. Interested landholders contact the authority, which sends over a field team. Training is offered to the farmer so that he may be able to recognise native vegetation and implement proper conservation measures. Based on two value scores, the field team assesses the native vegetation's quality. Interested landholders can submit several bids and specify, in a management plan drawn up with the field team, the type of vegetation and the conservation measures they are ready to undertake. In all, 98 farmers submitted 148 bids for 186 sites. 97 bids (offering the best value for money) were accepted, *i.e.* nearly 3,200 hectares of native vegetation for a total cost of about AUD 400,000 (Wunder *et al.*, 2008).

2.5.1. *When biodiversity can be produced like agricultural or forest products*

Of all the cases we have reviewed or studied directly, this is the one that fits most closely the model of PES as a direct, market-based arrangement. The only departure from the model is that the payer is the State – which appears consistent with the fact that this is one of many cases where what is bought is effectively biodiversity, a public good, for which a public "buyer" can be seen as the most relevant (Farley and Costanza, 2010).

Several factors play in favour of this arrangement operating satisfactorily here: (1) what is paid for is a certain vegetation cover, and administering conditional payments for given land covers is a routine practice of agricultural or forest policy and professionals, the relative ease of and experience in defining and controlling the conditions for payment being an important factor here; (2) the "sellers" each own areas of land that can provide the service irrespective of what happens in adjacent areas, and can deliver the desired land cover directly, through their own activity. In this case, there is no significant difference with the provision by individual producers of an agricultural or forestry commodity to a collective buyer; here the commodity is simply bought standing and is not removed after the auction. In other words, this case combines three characteristics that contribute to a satisfactory and market-based form of organisation: (i) the provision of the specific ES at stake is streamlined in the logic of the agricultural sector; (ii) ecosystem services are produced by individual, well-identifiable farmers; (iii) the Australian agricultural sector (by comparison with many other agricultural sectors, *e.g.* in Europe or in many developing countries) relies quite directly on market-based forms of organisation.

Such cases are the exception rather than the rule, however. Many, and probably most, situations where one intervenes for better ecosystem management differ from these on at least one of the three traits we just underlined.

(a) First, in many countries other than Australia, agriculture and forestry policies use economic instruments that do not make such an extensive use of competition among producers (or, one might say, that tend to protect the farmers from pure global competition). Farming subsidies in the EU or the US, development aid in the South, rely mostly on subsidy systems in which many considerations – like social support to farmers – are combined in technically and politically complex ways with the functioning of agricultural or forest produce markets, based both on political and technical considerations. Dealing with the beneficiaries of policies that strongly rely on subsidies is, again, very different from dealing with independent economic agents who produce and market agricultural or forest services, some of which qualify as ES, in a competitive way.

(b) In many cases, ecosystem services are not produced by individuals, either because there is some form of communal land management or because the services depend on a number of actors, none of whom alone can deliver the service (in the Vittel case, for instance, participation of all the farmers was a sine qua non for reaching the targets in terms of water quality). Both in theoretical and in practical terms, negotiating with communities or with collectives of actors is a different transaction than dealing with individuals on a competitive market (Pirarde *et al.,* 2009; Vatn, 2009).

(c) Last, many systems remain where conserving biodiversity and ecosystem services relies on land that is not managed by the farming or forestry sectors, or on land uses that are competing or in conflict with agricultural development or forest industry projects. We will come back to such issues in further chapters of the book. For now, suffice it to mention that it is an entirely different thing to give incentive to farmers or foresters for better ecological practices and to fund other sustainable land-uses so as to oppose agricultural or forestry projects that would alter them.

As we bridge the archetypal definition with practice we see real situations diverging from the archetypal case in different directions. This does not make PES irrelevant *a priori,* but it deeply changes their nature, according to the context in which they take place, and to the conditions they have to fulfil to make a useful difference. In particular, we would like to introduce and examine one specific issue relating to the "seller" of ES: are we speaking about a "producer", a "seller", or a "payee"? In a prototypical case like the "Bush tender program", these are one and the same operator, but in many situations it is not the case.

2.5.2. *What if the seller is not the producer?*

Seen from the most general perspective, biodiversity and ecosystem services are an emergent property of complex systems (Farley and Costanza, *ibid,* p. 2063). More precisely, they are part of the structure and results of the functioning of ecosystems and of the social systems with which these are usually deeply intertwined. In some cases, one can practically consider that one actor controls a tract of land supporting a self-standing ecosystem able to provide a given ecosystem service. The farmer in the Bush tender case is one example. The owner of a large tract of forest would be another. Cases where PES schemes are run on the premise that a given community controls a territory and ecosystem, and can be treated as the "provider" of its services, are also important in the PES literature. But it has been clear, from the early days of the environmental field and as we underlined above, that very often the conservation of given attributes of ecosystems relies on a heterogeneous set of social, political and economic actors, all playing a role in the existence of ES. Therefore, if "providing" is considered to be based on the control of land, and on decisions affecting land management, it becomes difficult to pin down who the "provider" of ecosystem services may be.

If we reframe the "payee" from producer to *seller,* the picture becomes somewhat different. As writes Wunder (2005, p.14) "*Whom exactly to pay is a question of negotiation, political feasibility (which includes perception of fairness), legality (particularly* vis-à-vis *land tenure) – and possibly also of ethics, since some actors may lose illegal revenues, corrupt payoffs, and iniquitous profits*.". This quotation captures in a nutshell three streams of thought in the literature, and their uneasy combination. The first is the pragmatic component of PES: the idea that by offering direct payments to some actors we may solve biodiversity conservation cases that have been intractable so far. The second is the concern over the social consequences of payments: are they going to benefit the poor? Or are they going to increase inequalities or create new ones (Pascual *et al.,* 2009; Pirard *et al.,* 2009; Sommerville *et al.,* 2010)? Is there a synergy between environmental and social outcomes, or are there trade-offs and dilemmas between the two (Landell-Mills and Porras 2002. ; Leimona 2009)? The third is the possibility for large-scale opportunistic behaviour, especially in systems where there are large public payments, based on property or rights that may have been acquired essentially for positioning oneself as the seller of services one does not produce (Pirard *et al.,* 2009).

Obviously, answers to these questions will be very different depending on cases and situations and on the leading principles one adopts for environmental and other policies. As the debate on PES now mostly stands, the issue is controversial and confused, as (1) it is highly charged because of the social and ideological tensions involved in these issues, (2) it is caught in tension between discussions of principle and examples taken from extremely different contexts, (3) it is caught between the archetypal, market model of PES, which is much too narrow to deal with these issues, and its sceptical critiques who tend to make sweeping generalisations on the negative effects of payments.

To sustain the effort for clear, context-differentiated design and assessment, there is a need for a better diagnostic of who is negotiating with whom for payments, and what contexts, strategies, processes and results of these negotiations may be decisive in terms of providing ecosystem services and of social consequences.

2.6. Paying is not only buying, it is interfering in a complex system

As we now turn from who is the seller to what does the payment buy, we may note that in the archetypal concept of PES, the payment buys a measurable, verifiable change in practice (for instance in land use) that creates improvements in a clearly defined ecosystem service. This change must be additional, *i.e.* one can separate changes that would not have occurred without PES and are thus ascribable to the PES. The connection between direct payments and clear effects is precisely where the archetypal concept of PES attracts both environmental economists (focused on efficiency) and pragmatic conservationists. The example given above, of conservation of birds' nests in Cambodia, would be emblematic of such straightforwardly tangible outcomes from a PES program: payments make interested villagers switch from egg and bird collection to conservation, for individually targeted payments.

But much of the literature acknowledges that this archetypical view is both far from actual practice in most cases, and that it also raises issues of concept and principle, so that both its relevance and legitimacy are challenged. Let us review a few of the issues raised by the literature, by practitioners we have interviewed and by previous case studies.

2.6.1. *The payment is part of a complex system from which a service may emerge*

PES are not – or only exceptionally – simple transactions. Not only do they require complex social negotiations, but the service itself is usually quite far from straight-forward. Again, they are often "an emergent property of complex systems" (Farley and Constanza, 2010, p. 2063). The presence of salmon in rivers results for instance from many factors: temperature of the water, presence of specific nesting habitat, complex migrations from oceans to rivers then from rivers to oceans, pollution, specific food, etc., all of them resulting themselves from various secondary factors such as the degree of erosion around the river, air temperature, etc. Those factors in turn are determined by anthropogenic drivers, which relate to a wide and complex institutional system. In other words, intervention on the (social-ecological) system from which the services emerge is anything but a straightforward process.

The archetypal definition sets payments in the context of buying a service. But in many situations, payments can be made in the context of arrangements that are not appropriately captured by the notion of "buying". Subsidies to rural communities and to small farms for a portfolio of diverse benefits that arise from sustaining rural social systems are a good example. Some services are indeed produced, for instance in terms of limiting migration to cities, of contributing to the maintenance of rural infrastructure, or of maintaining environmental services in terms of landscapes for instance. However, rather than being bought directly, they arise from a complex set of exchanges and policies, of which payments are just one component.

Another point relating to PES is that the nature of the change that is paid for generates confusion and practical difficulties.

2.6.2. *Paying for a "change" or for a "non-change": a source of confusion*

If we look closer at what PES are paying for, two kinds of cases can be firstly distinguished.

The first is a payment made to "freeze" (or conserve) some land use or practice, to restrict some uses. Without the payment, some practice would have been discontinued (for instance, grazing of flora-rich pastures) or introduced (for instance, addition of nitrogen to it). The second is a payment made to "branch off" on to a sustainable track, *i.e.* to support changes in practice that involve structural changes for instance in farming production systems (*e.g.* converting to organic production), or in long-term forestry options.

Wunder (2005), Pirard *et al.* (2009) rightfully underline a difference between the two situations. The first creates an unstable balance where unwanted changes are warded off provisionally. Here, paying for ecosystem services is more of an emergency or transitory measure than of a tool for change towards sustainability. It exposes the buyer to the liability of having to continue payments indefinitely and to the risk that the necessary amounts may rise if opportunity costs for the provider do. The second type of situation has much more potential for durability, because as payments push providers on a different route, their incentive structure changes. This is well illustrated by the Vittel case: once the changes in the farming systems have been obtained through the PES contracts, the technical and economic structure of the farm as well as its farm-industry context have changed in such ways that there is little attraction to revert to input-intensive farming.

The debate, however, is not so simple because in many cases in conservation, what is sought is an *absence of change* – for instance, not cutting a particular forest, not draining a wetland, not intensifying farming methods.

At a certain level of abstraction, "non-change" PES and "change" PES may converge: non-change can indeed be seen as change with regard to a baseline that would have involved changes. Nevertheless, this creates some confusion in debates: are we paying for change... or to prevent change that would have altered what we don't want to modify? This confusion creates difficulties in the implementation of PES: the rising use of counterfactual reference scenarios (what would have happened without the payment) as a basis for distributing hard cash does indeed raise great practical and fundamental problems.

Many authors criticise for instance PES system where the payment is based on very low opportunity cost as in a situation where poor farmers used to grow corn in an arid context. According to them it is the best way to be sure that these farmers will remain poor when otherwise they might have found another activity more profitable than the PES system (Karsenty *et al.,* 2010). At a global scale, the REDD+ negotiations also illustrate these difficulties: should the reference scenario be based on the past rate of deforestation in a given country, or on the expected rate?

The question of change or non-change is rife with theoretical ambiguities, as one tries to apply a simple linear form of reasoning (I am buying a well identifiable change) to what is, actually, a series of transformations in highly complex human and ecological systems. If we consider again the case of REDD+, the methodological problems are abysmal, since one is planning to establish baseline scenarios through foresight methodology, which is fundamentally conjectural (De Jouvenel, 1964; Mermet, 2009). Yet very concrete decisions and hard cash payments are to be implemented on that fragile basis. This confusion also creates opportunities for strategic manipulation, for instance when a would-be PES payee uses threats of negative changes in biodiversity to try and obtain payment for no action at all (Wunder, 2006; Gutman and Davidson, 2007).

2.6.3. *Focusing on the ecosystem service paid for may not be the best PES framing option*

A second important differentiation between cases on the basis of what the payment buys is the nature of the ecosystem services that are paid for. As one looks at numerous cases, the following facts become apparent.

First, a large contingent of examples, and often the most convincing ones in terms of their scale, robustness and financial flows, are about watershed management for water flow and water quality (Landell-Mills and Porras, 2002). In many of them (Vittel, for instance), biodiversity is hardly a factor.

Second, biodiversity conservation benefits from PES mostly in contexts where it is either a co-product or a by-product of payments for a wider bundle of services – usually involving water or carbon-storage as the main service. The distinction between a co-product and a by-product is important here. The first means joint funding of the scheme by a for-biodiversity-payer and a for-water-payer for instance, as in the "Los Negros" famous case, where two kinds of payments, one for water protection and one for birds habitats, were combined in the same scheme (Asquith *et al.,* 2008). The second is when a mechanism is motivated by water or carbon objectives, but also benefits for biodiversity, whether this side benefit is made explicit or not.

Third, it is most likely that the output on which the mechanism will focus will depend on who can pay. The vast majority of PES schemes are based on two types of services that correspond to two sets of buyers that are solvent and comparatively well organised: water consumers organisations, and organised recreational users (*i.e.* tourism, hunting, fishing operators or organisations). It is to be expected then that those interests will dominate in the arrangements, leaving biodiversity conservation that is not in their purview a "free rider" (Wunder *et al.,* 2008).

Overall, there is very little in the available cases that would support the idea that framing PES schemes in terms of ecosystems services significantly improves analytical clarity or provides better guidance for practice than specifying plainly what one is ready to pay for. If the economics of a scheme is based on paying for a given and specific product, such as bottling water in the case of Vittel, and the necessity this entails of nitrate-free underground water, the detour through the ecosystem services makes no real contribution to analysing how to get the benefits one is after. In other words, ecosystem services is a useful concept for general reflection and discussion on how we manage ecosystems, but has less value as an operational concept in the field.

2.6.4. *Is it rational to play chicken with social-ecological systems?*

If we consider the two kinds of PES, where a payment is made to "freeze" practices or to "branch off" to a sustainable track, what both types of situation have in common is that payments are supposed to be distributed to providers who are on the brink of changing practices. But in many field cases, payments are made to ES suppliers who are not in such a situation. As an example, Muradian *et al.,* (2009, p. 1206) cite the nationwide PES scheme in Mexico, "*where in some cases the government allocates payments for reforestation and forest conservation to peasant communities which hold forests in common* [...]. *This makes the directness of the transfer lower* [...] *and additionality is probably low* [...] *since indigenous communities will likely conserve the forests independently of the payment.* [...] *The very indirect transfer may be considered then a kind of reward for good environmental stewardship framed by rural development policies, instead of a market transaction between the State and rural communities.*"

The point we wish to underline here is that conservation of biodiversity and ecosystem services is a long run effort, where irreversibility and uncertainties often make it highly undesirable to be in situations on the brink of (often negative) environmental change. So one should take care not only of "freeze" or "branch off" situations, but also of situations where the main issue is to "consolidate" and stay far enough from the brink of change. This may not sit well with economic principles of additionality or marginality, but will undoubtedly, in many contexts, make perfect sense in terms of conservation strategy.

Overall, we are not convinced here by the theoretical objection that a payment for a stable situation would be irrational. To our mind the main question should not be: "Couldn't I pay a bit less and still not have the system go over the brink this year?". Once the complexity, uncertainty, time and spatial scales have been factored in, relying on such a theory may be equivalent to entering a dangerous game of chicken with a social-ecological system, by trying to test how close to the brink of sustainability we can go without taking action, and get away with it. As we mentioned above, a more reasonable framing of the question would be: "how is this payment part of a sound short-, medium- and long-term strategy to conserve the ecological items, structures or services we care about?"

2.7. PES in a wider picture: opportunities from synergy, risk of submission and opportunism

Framing PES as one tool amidst a composite strategy leads us once more to considering the tool for a larger trade, which is precisely the central question of this book. When connecting PES with other policy instruments and strategies, three topics arise from the literature and from field practice: the link between PES and legal obligations, the risks of seeing PES exploited and diverted, and the challenge involved in generalising and upscaling PES.

2.7.1. *Payments and legal obligations: are they synergistic tools?*

The link between PES and legal obligations is far from being obvious, as was evidenced in the case studies above. We could think indeed that PES are there to buy changes in practice only inasmuch as they extend beyond legal obligations, as is explicitly expected by people defending the principle of additionality. But this image is quite far from actual practice. In their review of PES in Costa-Rica, Daniels *et al.*, (2010, p. 2119) make this clear: "*A confounding issue regarding the additionality of PES in Costa Rica is that the law authorizing payments intended to internalize the benefits of ecosystem functioning also prohibits forest clearing (see Article 19, Ley 7575). One interpretation means that in the absence of payments, 89.1% of cumulative PES contract area, i.e. the total forest area conserved through the forest protection modality of PES, would have been conserved anyway via Article 19 if all landholders complied with the law. An alternative interpretation is that PES contracts serve as a necessary pre-condition for the application of Article 19 since the ban on forest clearing probably would not have been politically feasible without PES. As such, the ban on forest clearing and the effect of PES must be examined synergistically; it may be impossible to determine what the effects on forest cover are due to payments alone*". Far from being an exception, this situation is widespread (Engel *et al.*, 2003; Ferraro, 2008).

In our view, synergy is more relevant as a principle than additionality to account for the link between PES and legal tools (and indeed also with the other tools in the conservation panoply). In our case, assessing synergy means assessing whether a strategy that uses PES along with other instruments does better than without PES, in terms of environmental results. One might argue that this is similar to additionality, since it involves measuring the difference between outcomes from a strategy with and without PES, and attribute the difference – the additional effect – to PES. But this synergistic effect is quite different from the principle of paying for those changes that go beyond legal obligation. As suggested in Daniels *et al*'s analysis of the Costa Rican case, the positive outcome of the payment may indeed come from the fact that a monetary transfer has made possible a change in the legal framework, for instance by making it politically palatable or by making its enforcement more feasible on the ground. As a basis for making payments, this is a completely different model than the direct purchase of a specific non-mandatory service as suggested by the search for "pure" additionality. It is, however, consistent with and comes in support of the wider definition of PES proposed by Muradian *et al.* (2009, p. 1205) as "*a transfer of resources between social actors, which aims to create incentives to align individual and/or collective land use decisions with the social interest in the management of natural resources.*"

2.7.2. *A potential for synergy, a risk of opportunistic exploitation*

Furthermore, the approach of basing evaluation of PES on a comparison between the same strategy with and without PES is questionable if one of the options is not feasible. So we ought also to include in the comparison not only the same strategy with and without PES, but other strategies as well.

When a reference strategy fares better with PES than without, this does not rule out the possibility that some other strategy would fare better still than either of the two. This is highly relevant both to the theory and practice of dealing with the biodiversity crisis. The success of ETBs should not shrink the frame to considering only existing conservation strategies, complemented or not with ETBs such as PES. This would contribute to locking ourselves up in existing trajectories, even if unsustainable.

Telling apart "additionality" and "synergy" of PES leads us to the heart of the overall approach to the use of ETBs that we are advocating here. We do concur with the many authors who have shown that, in practice, PES function in very different ways than pictured by the archetypal model. This does not lead us, however, to seeing them, as the more radical critics would, as groundless and generally ineffective or perverse. We would rather support those who advocate investigating in-depth the practice of economic tools. Our main point, however, is that we have to become able to frame the discussion of that practice in a systematic and theoretically informed way. Above all, it means considering the overall options and strategies for biodiversity, of which PES, and ETBs in general, are components. Whereas the "additionality" of PES reflects the framing of micro-economics, its limitations suggest replacing it by the more extensive concept of "synergy". But the analysis and practice of the synergies that can make PES useful requires not only a pragmatic discussion (which the PES literature shows is well underway), but also to be founded in explicit and appropriate visions of biodiversity policy, management or governance. "Synergy" here means that a given tool works in combination with other tools in the wider framework of a given course of action. Biodiversity researchers should focus their attention on these combinations and these wider frameworks.

It should be stressed, however, that the concept of "synergy" suggests a framing for analysis, not a solution. Just like "cooperation" (the exact Latin equivalent to the Greek etymology of "synergy"), it may be taken as suggesting that a joint operation of tools tends to produce positive effects. But in the field of PES, concurring with our own research on environmental management, there is a whole stream of discussion in the literature and in field experience that shows this not to be necessarily the case (Mermet, 2011).

Let us consider a case from France – a large public program with considerable payments and limited environmental effects – that could be seen as symmetrical to the Vittel experience.

The "Program for the Limitation of Pollution of Animal Origin" (PMPOA being the French acronym) was initiated in 1993. Its principle is to offer payment to farmers who undertake to upgrade their buildings so as to limits leakage of manure, and to improve manure storage so that it can be spread on fields only at times where it has less negative environmental impacts (essentially, to try and keep pollution at legal levels). Over its first period (until 2001), the program was allocated the equivalent of two billion euros of public funds. In his very detailed study of the implementation of the program in part of South-Western France, Didier Busca (2010) shows how it involved multiple successive negotiations. Each of these negotiations played on the necessity for the authority to obtain the collaboration of other actors without whom implementation could not proceed. Consider one example out of the whole set given by the author. The farmers' union, without which hardly any farming policy could have been implemented on the ground, required that the subsidies be available too for new farms, to set up young farmers. The authorities did not manage to refuse this because, in terms of rural development, settling young farmers is a priority. The result is that part of the funds goes not to help older farm buildings reduce their impacts, but to subsidising new farm buildings that conform to the law. More generally, Busca showed that the entire sequence of such negotiations involved in implementing the PMPOA in the region he studied resulted in practically suppressing all environmental conditions that had been included in the initial design. Besides, the positive environmental effects of this programme have not been demonstrated yet, as witnessed by France's persistent breaches to the Nitrates directive objectives, and continuous pollution of Breton beaches by green algae.

This program is obviously not an archetypal form of PES, but – at least in its initial design – it does qualifies as "a transfer of resources between social actors, which aims to create incentives to align individual and/or collective land use [here, rather farming systems] decisions with the social interest in the management of natural resources" (Muradian *et al.*, 2009). It also illustrates the same sort of synergy with law presented in the analysis of Costa Rican PES (Daniels *et al.*, 2010). The payments from the program do operate in complex combinations with other economic, legal and administrative tools; and they operate within the system of actors that benefits from the transfer of resources. In this case, the PMPOA "works with" the overall farming subsidy system, technical support system, administration and politics. The net result can hardly be seen as a positive "synergy", as the resources given by society to improve the environmental situation have mostly been appropriated by the farming sector as additional general subsidies to farmers, without environmental benefits. Busca and Salles (2002) proposed the concept of "eco-opportunism" for such situations where as new resources intended for conservation are deployed, the real conditions of their joined deployment with other actors or policies are in fact voiding them of their conservation intentions and conditions. In other words, the payees opportunistically seize the opportunity and exploit the payers by not delivering.

2.7.3. *Synergy between various environmental goals of policies cannot be taken for granted either*

Similar issues may arise between various components of environmental policies. For instance, wetland restoration and conservation is advocated for multiple reasons that include mitigation of urban flooding and water quality. This is usually presented as a synergistic situation, where if you obtain wetland conservation, for instance through compensation payments to farmers combined with urban planning, you will reap both the benefits of flood mitigation and biodiversity conservation. But this is not the case if the wetland protection approach serves one of the objectives much more than the others – for instance if the payments to farmers are based only on keeping pastures so that flood expansion is not a major problem, but with no conditions on intensification of the pastures, therefore with little effect on water quality, not to mention biodiversity. In his book investigating how water issues can be managed with respect to their spatial dimension, Narcy (2004) shows that in many cases, relations where environment and development policies are presented as "synergistic" are in fact situations where biodiversity objectives are dominated by other priorities.

Table 4 *Four positions of biodiversity policies vis-à-vis other policies*

	Effects of environmental and other policies are divergent	**Effects of environmental and other policies are synergic**
Passive position of biodiversity policies vis-à-vis other policies	Biodiversity policies are *troublesome*	Biodiversity policies are *submissive*
Active position of biodiversity policies vis-à-vis other policies	Biodiversity policies are *conquerors*	Biodiversity policies are *opportunists*

Source: Adapted from Narcy 2004.

Such processes of "eco-opportunism" and of "submission" of biodiversity to other policies are a central concern in the current development of PES. Take REDD+ for example. There is already the risk that part of the funds end up going to the forest sector without tangible environmental returns, through an eco-opportunistic process. But there is also the very real possibility that even if it delivers environmental benefits, those would be only in terms of greenhouse gases, with payments being based on carbon-storage services, and that biodiversity become a collateral damage (for instance if the payments go to biodiversity-harming plantations), or a co-product (if conserving some forests for carbon also conserves them for biodiversity). It is interesting to see that in much of the debate on REDD+ and biodiversity, the latter case is framed as biodiversity being a "free-rider". But this perspective is only that of the for-carbon payer. From the point of view of biodiversity concerns, the situation is better seen as one of "submission" of biodiversity to other concerns and policies, either environmental, as in the case of carbon in the context of REDD+, or more generally, as is the case with rural development in many other biodiversity conservation contexts on the ground.

As one moves beyond the simple model of additionality to look into synergies, opportunism and submission, it becomes quite clear that we have to enable ourselves to analyse, for each case, what is the place, role and effect of PES in the overall interaction of actors, strategies and tools that will, or will not, deliver the outcomes that the ecosystem-services payer expects.

2.7.4 Upscaling and generalising PES: potential and pitfalls

Such an analysis will not be necessary only on a case-by-case basis, however. It also has to inform reflection on the issues involved in the potential upscaling of PES. This topic is of great importance, if one takes the "promise" of PES to be a potential solution for the biodiversity crisis. However, as Pirard *et al.* (2009, p. 255) warn: "*Programs that promote the extension of PES on a large scale through the replication of the original blueprint must be regarded with caution.*" These authors insist that we should be wary of "*the belief that the proliferation of PES schemes is a solution to the current problem of massive environmental degradation.*" (*ibid*) Let us take stock briefly of the difficulties they raise.

First, from an economic standpoint, if PES manage to divert significant areas of land away from the most productive forms of use, this may increase the opportunity cost PES would need to base their payments on (because land for productive uses will become scarcer).

Second, if land for productive uses becomes scarcer then it may decrease overall well-being because of the limitation of production capacity (Karsenty and Nasi, 2004).

Third, if one looks at PES from the point of view of the funding of environmental policy, upscaling PES can lead to increased tensions with environmental spending through other channels and for other environmental purposes.

Fourth, if one focuses more on social actors and their interactions, the flow of payments involved in PES can, as we have seen, be expected to trigger strategic behaviour from those who expect to benefit from the payment. The point here is that, as the amount of funds involved increases with upscaling, the incentive and the opportunities increase for actors who are motivated by the possibility to create rents for themselves through opportunistic or exploitative tactics. As the scale changes, the power problems increase and change in nature. In the rent-seeking process, which exists both in developed and developing countries, trying to act on the biodiversity crisis through large-scale payments raises specific and formidable challenges. As we know, important environment-impacting sectors (like agriculture or forestry) are also important rural and economic sectors. They have proved quite successful in appropriating large-scale public funding while often retaining a very reticent position regarding biodiversity conservation goals.

Fifth, as one upscales, one has to move from the consideration and treatment of local issues through PES to the wider picture. Then, it becomes no longer possible, in the words of Pirard *et al.*, to neglect the political choices regarding development trajectories: when a local approach is envisaged at nationwide scale, it necessarily interferes with many other social issues and political priorities.

Payments are made on the basis of individual transactions between a payer and a payee, and this "de-centralised" nature is a central part of the PES concept and innovative potential. However generalisation and upscaling naturally challenges this very foundation of what gives PES its appeal, both pragmatic and theoretical, as a decentralising solution.

We find striking, as we examine PES cases, that they often seem to have succeeded precisely because they are dealt with as exceptions. This is clearly the case for Vittel, where the farmers' union has resisted and finally tolerated (but not approved) the PES scheme because it did not want it to constitute a precedent for a change in production techniques. Researchers who have analysed the case clearly show to what extent and for what reasons it constitutes an exception in the landscape of French policies to limit the impact of farming practices on water (Perrot-Maître and Davis, 2001). If PES schemes are negotiated exceptions, "islands" of sustainability in a rising sea of (largely policy-backed and funded) unsustainable development trajectories, one should not take for granted an upscaling based on multiplying approaches that are founded on pilot experiments, *i.e.* on a generalisation of exceptions.

To sum up, what is needed is a diagnostic of how PES fit in the overall picture of the (ecological, social, economic) dynamics that may deliver the expected biodiversity outcomes. That picture has to be analysed at the relevant scale. If one wants to upscale expectations from PES – be it from a spatial, financial or policy point of view – one has to support that ambition and analyse the social, political and economic systems involved at the appropriate scale.

Conclusion

After discussing all the components of the archetypal definition of PES – the buyer, the directness of the deal, the seller, what is being paid for – the discrepancy between the model of a simple market transaction, and the reality of a contingent negotiation in complex eco-socio-systems, is more apparent than ever.

One emerging conclusion is the need to characterise and analyse more clearly such situations and negotiation processes – a need we will address in chapters 5 to 10.

Moreover, whereas the model of a straightforward transaction is the exception rather than the rule, and PES are not exempt of the many challenges of environmental policies in general and biodiversity conservation in particular, it does not follow that they do not entail an innovation and a specific contribution to the conservation toolbox. Even in complex and contingent situations, they function on the combination of three principles: (1) enrolling the "contributing" potential of beneficiaries from ecosystem services, (2) seeking more direct, or at least, new relations between economic agents on either side of activities with negative or positive environmental impacts and (3) seeking leverage entirely based on incentives and on volunteering by producers of positive externalities (Laurans *et al.*, 2012). This combination of principles does have a high pragmatic appeal and can complement other environmental action tools in a variety of contexts.

What also emerges from the great variety of examples available is that, far from breaking away from other, "traditional" environmental policy instruments, PES are fundamentally complementary to them and dependant on them. The synergy between law and PES in the Costa Rican case has provided a large-scale example in a complex context. PES just cannot be seen as an alternative to the other options in the environmental toolbox. They are always used in combination with other tools. And within the financial dimension of environmental management and more generally public policy itself, this interdependence with other instruments is decisive. PES are in most situations added to a whole web of pre-existing transfers that are constitutive of the economy on which they intervene: subsidies, aids of various kinds, exemptions, in-kind support, etc. A PES is almost never a money transfer that comes in what would otherwise be a completely free-market context. As one proposes to pay farmers for a change in farming practice, one has to take into account this new payment will just be the last in a long list of payments they are already receiving, for instance to replace grassland by maize, to install irrigation systems, or to replace forests by pastures – all those examples being harmful payments for biodiversity. Again, this is not an argument against the use of PES as a tool in general, but rather to advocate that PES may be of limited use if, as we design PES schemes, we do not also revisit the coherence of policies affecting the environment and of the monetary transfers they already involve (Vatn, 2009).

3. Buying land or land-based rights for conservation: ownership is just the beginning

Introduction

After ecosystem services valuation, which may exert influence on biodiversity quite indirectly, through an impact on the policy process, and PES by which someone tries to modify someone else's practices by changing the system of incentives that drive their behaviour, we are now turning to a set of ETBs that are deemed most direct. They consist in buying land or land-use rights directly on the market, for conservation. What could be considered more plainly "economic", or market-based than such a direct purchase?

The importance of tools based on buying land or land-use rights is easy to grasp if one bears in mind that biodiversity is highly dependent on land-use and that many biodiversity components need permanence in land-use.

In this chapter we will first describe, define and exemplify this family of ETBs. Then we will screen both some of the theoretical arguments proposed in favour of them as well as the most common "critiques on principles" that are dealt with by the academic literature. It will then be possible to dwell upon the core of the chapter, analysing these ETBs "in use". Conversely to an approach that would see "buying conservation" as a once-and-for-all solitary and purely private tool, we will find that this set of tools actually needs an alignment of goodwill, a long-lasting negotiation capacity, and a strong dependency towards surrounding administrative, legal and political interventions. In so doing, we will illustrate how analysing the use of these ETBs from a strategic point of view gives birth to perspectives that may effectively complement the critiques that focus on their principles. We will also see how this contributes, in a specific way, to pointing at the need for approaches to strategy, which will be the subject of the final chapters of the book.

3.1. The 'Buying for Conservation' family of ETBs

3.1.1. *Land acquisition*

The first and apparently most straightforward ETB of this family is purchasing land outright for conservation goals. In theory, this can be done by whoever pursues conservation objectives and wishes to invest his funds to that purpose.

This probably exists widely. As underlined by Carter *et al.* (2008), nature reserves created by private persons have indeed long been important in European conservation. In France for instance, after his 1950s purchase of the Tour du Valat estate in Camargue (South of France), Luc Hoffman established an important nature reserve and a renowned conservation and research organisation. The conservation of biodiversity on private lands was not only important in Europe (Kleijn and Sutherland, 2003; Rafa, 2005) but also in the USA (Bernstein and Mitchell, 2005; Newburn *et al.*, 2005), Australia (Figgis *et al.*, 2005), and in many developing countries (Langholz and Lassoie, 2001; Langholz, 2002; Adams, 2004). A major reason for this was the growth of the global nature-based tourism industry, wildlife-based photo tourism and recreational hunting (Christiansen *et al.*, 2005). In southern Africa a substantial game ranching industry developed on former cattle ranches, based on safari hunting and photo tourism enterprises (Suzuki, 2001; Wels, 2004). Wealthy individuals also purchased land to establish private reserves or conservancies (Chudy, 2006).

However, most of the visibility of biodiversity conservation through the purchase of land or land rights is linked to land purchase by non-profit private organisations like trust funds and foundations. Such interventions are well known to the public and the literature; their large size and publicity make them visible. In the second half of the 20th century landholdings by conservation trusts in a number of industrialised countries grew substantially; for example, the Royal Society for the Protection of Birds and the National Trust in the UK (Tunbridge, 1981; Dwyer and Hodge, 1996). In the USA, the Nature Conservancy (TNC) is a good example of how private trust funds may operate by buying land for conservation. Founded in 1954, and today with over one million members in the United States, TNC claims to pursue '*non-confrontational, pragmatic, market-based solutions'* to environmental challenges' (TNC, 2011). Through purchases and gifts, TNC has acquired over 15 million acres of land in the United States (6 million hectares). In recent years, TNC has also begun to fulfil a number of functions analogous to a market-maker or broker in respect of the purchase of private lands. TNC will first buy land that it considers highly valuable for its biodiversity. It will then seek a private sector buyer committed to preserving the land, and will either partner with this buyer or sell the property on to him. In Australia, the NGO "Australian Bush Heritage Trust" (ABHT) identifies and purchases for example high conservation value land, and has also pioneered the use of conservation easements.

The same actions are also undertaken by state agencies in pursuit of conservation and biodiversity protection. For example, in France, the Conservatoire du littoral was established in 1975 to purchase and permanently protect coastal lands. Once acquired, management is entrusted to local municipalities or regional organisations. By 2010, 135,000 hectares had been acquired, with the goal of ultimately protecting one-third of the French coastline.

3.1.2. *Conservation Easements*

A conservation easement is a legal restriction placed on a particular property, to limit development activity on it or restrict the use of its resources in some way. More specifically, "*A conservation easement is a legally binding agreement between the owner of the land encumbered by the easement and the holder of the easement that restricts the development and use of the land to achieve certain conservation goals, such as the preservation of wildlife habitat, open space, or agricultural land*" (McLaughlin, 2005).

Trust funds such as TNC, which buy land for conservation, also participate in the use of easements. For example, land may be purchased, and a conservation easement imposed on it, following which the land may be sold to a third party. The Trust continues to hold (and if necessary, enforce) the conservation easement and through selling the land gets additional funds available to invest in new land purchases. A similar strategy is enacted by the Bush Heritage Trust in Australia.

A good example from the South is provided by the Lake Yojoa conservation easement in Honduras.

To improve the condition of Lake Yojoa, The Nature Conservancy worked closely with an association of mayors which was formed in reaction to perceived threats and degradation of the lake's condition. During a first phase, they produced a participatory plan: collectively valuating the benefits from the lake, defining priority areas, designing conservation plans, etc. This was done based on an extensive participation of stakeholders, through workshops. Among the actions that came to force subsequently, five conservation easements were signed, with technical assistance from TNC. To allow and foster this initiative, the Honduran government decided to exempt from property taxes all lands under easement, and to assist them for fire protection where needed. The mayor association provided legal assistance to the landowners to obtain legal rights or official title of their land. Subsequently, however, serious difficulties appeared in implementation, especially in: land disputes and claims on properties as well as infringement of the easement conservation provisions. TNC acknowledges that the weak property system, and the high and poorly met staff needs to monitor compliance are hindering factors for conservation easements in such countries (Krchnak 2007).

Another way these trust funds apply easements is through negotiating with individual land owners. TNC will purchase an easement from an owner, with the land remaining in private hands and often the existing use continuing. However, any future sale of the land will occur subject to the easement. Any change in land-use will also need to comply with the easement.

Some public bodies use the economic tool of auctions to ensure they may purchase the greatest total easement restrictions for their budget. For example, IUCN reports that local counties in Colorado, in seeking to limit urban sprawl, have set up dedicated budgets for the annual purchase of conservation easements in designated 'low development' zones. The county calls for bids and will then purchase easements it considers to be of the greatest long-term ecological and environmental value.

3.1.3. *Conservation Concessions*

Conservation concessions are a much newer form of tool. It consists in purchasing land-based rights not for eternity, but for the long term. It follows the pattern of the concessions through which States sell rights of access to resources they control for mining, farming or forest exploitation.

A first form of conservation concession is simply the purchase on the market of land-based exploitation rights by a buyer who does not intend to exercise that right, but will rather conserve the resource. The prototypical example would be the purchase of a forest logging concession for non-logging. Examples are very rare. In the northern hemisphere, there is the famous port of Orgambideska in the Pyrénées, which is worth a summary here. In the Pyrénées, local authorities auction nine-year concessions for hunting migratory birds on ports that are part of the communal estate. In the 1970s, to everyone's surprise, a bird protection NGO enters and wins the auction. The port will be hunt-free for the next nine years, and used for counting migratory birds. The anger of the hunters associations is great. In 1982, the communal authorities set a fixed price for the port that was too high for the NGO. The sale failed. A few months later, the communal authorities sold the rights to the hunters association for half the price. It is interesting to note that this – bidding for concessions – may be the most clearly market-based instrument examined in this whole book:

the conservation of a given biodiversity item directly competing on the market with an activity that degrades or destroys it. Even more interesting is the fact that it raises fierce criticism, as if conservation was challenged to go on the market… in all cases except those where it may win. Beyond the paradox, this observation points once again to the limits of the autonomy of economics in the management of natural resources.

A second form of concession is an adaptation of the principal of the resource concession, purposefully directed to conservation by the authorities. They are then management contracts between a government or community landowner and a conservation-minded buyer. Unlike a park, a concession reaps revenues, making it more appealing to host governments. And unlike a park, or an easement, which can lock up land forever, a concession is temporary, albeit renewable (Ellison, 2003). Here are two examples from the South.

A conservation concession agreement was signed in 2001 for 40 years in Peru where the government specifically allowed conservation concessions in its new Forestry and Wildlife Law, passed in 2000. The land in question is a 135,000-hectare forest in the Los Amigos watershed, in Madre de Dios province, adjacent to a national park. It is managed by the Amazon Conservation Association, a Non-profit NGO incorporated in the United States and Peru. This NGO lobbied strongly for a conservation concession on the Los Amigos site because it was already developing a research facility on property it owns outside the concession boundaries. It has since agreed to invest US$5 million in infrastructure, salaries, and conservation management expenses over the first five years of the concession. It also created a US$1-million endowment to pay salaries for 11 rangers over the term of the contract

In southern Guyana, another conservation concession agreement was signed in July of 2002 between Conservation International and the Guyanese government to lease timber rights to 80,000 hectares of pristine rainforest alongside the Essequibo River. The agreement involves payment of acreage fees and royalties comparable to an active timber concession. This site was not facing any immediate threats from loggers but it seems that such threats were bound to increase as neighbouring Brazil continued to build roads improving access to the region.

It has to be mentioned that both of these concessions are in remote and all but unpopulated areas, and in neither case did the conservationists face competitors for their bids.

Now that we have defined and illustrated this family of ETBs, we will briefly review how they are debated in the literature, looking especially at the critiques they are generating.

3.2. Rationale and critiques

3.2.1. *Private property arguments disconnected from pragmatic rationales*

Both economists and environmentalists have argued in favor of these instruments.

One stream of thought pleading for using land property as a tool for conservation is rooted in theories that assign responsibility for environmental problems to the excessive power of governments, and seek the solution in rolling back government intervention and handing responsibility over to private property. The following quotation from the economist Walter Block (1990, p.281) illustrates the kind of arguments that are frequently used to defend this position: "*There is a well-known expression to the effect that everybody's business is nobody's business. This applies with particular force to the issue of pollution. There is a problem because the air, the water and the forest that are polluted are in most cases everybody's property* (…). *What that means is that they belong to nobody. Therefore, no particular person objects when land is infringed by a polluter or when trees are killed by airborne noxious chemicals*". To continue with this same author the following quotation explains how this covers biodiversity too (p.281): "*The question is how do we ensure that the protective reactions of private property ownerships will leap to the aid of the forest* (…)*? The answer is that as long as we persist in the myth of public ownership, it will be very difficult*".

Environmentalists too have been defending these instruments for long, acknowledging the importance of the private sector to conservation strategies in the 21st century (Langholz and Lassoie, 2001; Bernstein and Mitchell, 2005). The Convention on Biological Diversity's programme of work on protected areas comments for instance on the importance of recognising and promoting 'a broad set of protected area governance types [including] private nature reserves' (CBD, 2006, Prog. Element 2: Specific Goal 2.1). To face the emergency of the biological crisis, environmentalists sometimes defend that private actions and decisions are quicker to take and implement than public ones. An member of the NGO Conservation International explains for example that conservation concessions can be used to conserve large areas of land over an entire region as a temporary measure until a formal network of protected areas can be planned and implemented [17].

[17] http://www.ecnc.org/file_handler/documents/original/view/258/conservation-concessions--concept-descriptionpdf.pdf

Compared with the other types of tools we are reviewing, the gap between theory and practice is much larger here. In chapters 1 and 2, we have seen that between the economic theory rationales for ESV and PES respectively and actual practice there are considerable tensions and discrepancies, but there is an active and productive dialogue. In chapter 4, we shall find that to be the case too with biodiversity banking. Here, however, there is very little connection between the promotion of private property as the foundation for sound environmental management on the one hand, and the actual use of land and land-use rights buying for conservation on the other. One reason may be that the bulk of cases where the tool is used is constituted by purchases by collective non-profit or by state operators. The cases in actual practice that could actually mirror the "salvation-in-private-property" theoretical arguments are those where private owners privately buy land and privately conserve it. Such practice is massively under-documented. It surfaces only when the owners turn to the State to ask for subsidies or tax breaks associated with conserving the land. But by doing so, they step out of the theoretical framing that state intervention is counterproductive... A second reason is that the notion of an economic tool, or an economic instrument, does not really apply to an approach where individual economic agents buy what they want (here, conservation) without any collectively organised intervention or policy.

As the field stands now, (1) we can only call for more research on situations where private operators buy or hold property for conservation. For the rest (2) we shall consider that the main argument of principle currently and effectively underlying the use of land purchase as a tool is the pragmatic one: buying land (or leasing long term concessions) is an expedient and trustable tool for collectively organised intervention in favour of conservation, whether the operator has private legal status (NGO, trust-fund, foundation) or is a state operator.

3.2.2. *Tools based on purchasing land-based rights trigger vocal critiques*

In this perspective, even if we leave aside the numerous critiques of arguments that promote private property versus government intervention, it is interesting to see that the pragmatic argument in favour of tools based on purchasing land-rights also triggers a specific barrage of critiques. These target respectively five issues: (1) demeaning the morality of making a living through work that relies on the exploitation of natural resources; (2) the inequitable sharing of biodiversity endowments and their benefits; (3) the risk of excessive power vested in private conservation actors, (4) the discrepancy between what would be scientifically justified for conservation, and (5) the pragmatic demands of real-estate deals.

- The moral critique. These tools have been criticised on a moral basis for equating salaried work with conservation payments, thus described as "rents" and criticised for their sterility and their tendency to reduce the possibility of development (Karsenty, 2007). This argument is particularly vivid when the purchase is designed to subtract a given land from a projected intensification of exploitation, as when an environmental fund competes with other auctioneers to buy an exploitation concession and use it for preserving the environment. It may be inferred from this type of comment that those who make it are critical of placing impediments on the extraction of natural resources as a means of development, and that they see industrial development, agricultural intensification or urban development as moral imperatives. This critique seems to be grounded in the moral attachments of the authors to certain kinds of productive labour, for instance in industrial agriculture or in forest exploitation. How else can their dismay at the thought that a worker may be remunerated for caring for part of an ecosystem, against their approval that he might be remunerated for cutting down trees and hauling them to a timber mill be explained?
- The equity critique. This argument is that allowing some communities to prosper from biodiversity stewardship whilst others, lacking biodiversity endowments, are denied income which transient labour may provide through development of forestry and agriculture, is inequitable (Karsenty, 2007). However, the uneven distribution of biodiversity is inherent to the very concept, and such a critique appears somewhat trite. Biodiversity endowments are no more unequally distributed than other natural resources such as oil, minerals or fish stocks. Yet the argument is not heard from these authors that forest exploitation should be prevented because it may unequally benefit populations close to the forests, relative to those in areas with no forest resources. Political ecologists have also made the critique that local populations, which may rely on biodiversity for their livelihoods, are often excluded from traditional lands as a result of conservation measures enacted by the purchase of land (Mahoney, 1992). This is the same argument that is often made against protecting areas through regulation, which may also result in the permanent exclusion of some users. The important issue here should rather be the exact nature and degree of limitations put on use by the holders of rights, and the potential shift in access rights among various types of users. This can be exemplified by the recent acquisition of former industrial salt marshes in Camargue (France) by the Conservatoire du Littoral. These salt pans were previously closed to the public, but some local users were allowed access for hunting and fishing. The Conservatoire's mandate

includes a duty to open land to the public at large, whereas the use of public money ear-marked for conservation is not compatible with some former hunting or fishing activities. This change in ownership and shift of populations that are entitled to access is currently generating local conflicts.

- The governance critique. The fact that this tool relies on (and generates) large property-holding organisations, handling sizeable amounts of capital, is also subject to criticism. For instance, as trust funds have grown in scale and scope, they are considered as having taken on the political importance of major institutions. Their governance regimes have been criticised for being too complex, and their purported self-interest has been challenged. Articles in the *Washington Post* alleged that TNC did not criticise business interests allied to its board members, and had engaged in questionable lending practices to related parties (Ottaway and Stephens, 2003); (Stephens and Ottaway, 2003). These criticisms may in part be well founded. The issue, however, is to establish in what measure critiques have a problem with conservation organisations handling large estates, with the associated level of power and wealth, or if they engage the way they exercise this power and wealth.
- A further criticism involves the subjugation of scientific and ecological priorities to the necessary parameters of property transactions. '*Science is not understood or supported by senior managers and state directors. [Their] entire focus is on land deals*' claims one scientist from TNC, quoted by the Washington Post.

What strikes us with these critiques is that none of them addresses the specific issues of land-rights purchase as a tool. The first three seem to be concerned only with the fact that the conservation effectiveness of the tool may hamper some aspect of sector-based (e.g. forestry, agricultural) development. They respectively challenge the idea that conservation (rather than lumberjacking, for example) can be a source of revenue, that high biodiversity can be an asset to certain communities and that some conservation operators can gain substantial power. In a way, these three arguments find fault with the tool because it can work; they are indeed an answer to the pragmatic rationale in favor of the tool.

As for the fourth argument, it blames land-rights purchasing instruments for what is in fact valid for all biodiversity conservation instruments: all tools, whether economic or regulatory, rely to some extent on managerial, political or market opportunity. No policy instrument is or has ever been based solely on science and technical optimisation. No one would expect a public scheme to be implemented with no consideration for economic, regulatory, social, symbolic, security considerations. The issue is one of striking a balance between the practical constraints of what are in effect real-estate situations and the biodiversity science involved in the goals of a conservation intervention.

Overall, whereas the debate on principle is just as lively here as for the other types of ETBs, it is less connected with and relevant for practical realities. Still, we need to address the practical issues of using this family of tools. What caveats and suggestions can be drawn from an analysis of the practice? Not much literature is available on the subject, so we will have to dwell more upon our own experience and resources.

3.3. Challenges met in practical use

3.3.1. *The importance of goodwill*

In most of these ETBs, landowners are primarily motivated by the possibility to value environmental qualities of their lands, along with their private interest when this allows tax exemptions. Their action is voluntary and relies on their individual initiative. Boyd *et al.* (1999) consider that part of their success comes from the fact that they are "more politically palatable", because they rely on voluntary rather than on constrained behavior. Second, the conservation organisations generally act on behalf of donors, be they private or public. Last, the whole institutional context is set up by strong policy acts, and then by strong political leadership. But the needed political support is not once-and-for-all: the legislation has to be adapted through a national Act, it has then to be facilitated, relayed, implemented and developed at local scale (local government) by equivalent legal provisions. It has also to be adapted from time to time to correct observed diversions and perverse incentives; it has to be defended politically against critics. As shown by both the Conservatoire du littoral in France, and the various private trusts in the US, the cause of conservation easements and purchase of land for conservation is very popular with the public. Historically, this support has emerged from a desire for the preservation of areas of particular scenic beauty (McLaughlin, 2005) but it now tends to also apply to areas of high biodiversity value.

In other words, *goodwill* is critical in voluntary conservation transactions. There has to be goodwill from the funder of the purchase and from the sellers of land or easements. The authorities also have to be well-inclined to such conservation to put into place the appropriate legal frameworks. This may be one of the reasons this family of tools stands out from conservation tools that use an element of regulatory compulsion. These tools are inherently grounded in the public's willingness to fund and in the support of the local stakeholders. Therefore successful strategies will usually be based on non-aggressive action and the investment of time and energy to create a shared vision. They are often presented as offering a way forward to defuse environmental conflict.

The fixation of prices for the purchase of property rights also tends to rely on voluntary agreements. Parties may be directed through legislation to consider the opportunity cost of land uses forgone. In the US, federal and state laws define the basis on which remuneration is justified, as in the following US Dept. of Justice provision: "*if the property is clearly adaptable to a use other than the existing use, its marketable potential for such use should be considered in determining the property's fair market value.*" (cited by Boyd et al., 1999). Opportunity costs, however, rather seem to frame the commencement of pricing discussions, and perhaps provide a theoretical ground on which bargaining may occur, but it is supply and demand which set prices. Again, in all cases, a successful transaction requires willingness on both sides.

3.3.2. *A wide scope of operation*

Tools relying on buying property rights have been used successfully in very contrasting contexts, especially regarding the value of land and pressure for land use.

A first type of situation is quite similar to the one identified for the success of PES in general: cases where the target is to maintain an extensive use (typically, ranching) that is favourable to biodiversity and other environmental qualities. This is for instance the case of buying easements on lands under moderate pressure for urbanisation or under pressure of agricultural intensification. In that case, large surfaces can be protected at a manageable cost.

But, contrary to what is the case with PES in general, purchase of rights also prospers in much more extreme situations. At one end of the spectrum, it is largely employed in areas where pressure for urbanisation is very strong – including to keep green areas in the periphery of cities. The reason is probably that under such pressures no other tool than buying the land, or buying building rights away, is able to ensure continued biodiversity friendly use, or restoration. As a consequence, land buying is used even when costs are very high, but balanced, however, by very high public demand and capacity to pay (as for scenic landscapes, charismatic species, and green spaces in dense urban contexts). The example of the Conservatoire du Littoral buying high-real-estate-value land on the sea front also illustrates this sort of situation.

At the other hand of the spectrum, in remote areas, or in poor countries, extensive tracts of land, or restrictions of uses, may be purchased for comparatively lesser costs. This opportunity effect, similar to the opportunity of creating protected areas in remote regions, can make the acquisition of rights very effective and efficient in terms of conservation, especially if it allows anticipating by buying rights while they are still cheap. More than one important ownership-based conservation area in Europe is the present result of such strategies in the past.

Finally, it should be noticed here that this family of ETB appears as the only one that can operate in contexts where strong economic drivers are to be counterbalanced in order to protect biodiversity. Other ETBs, because they try to do much with little money, and because they act indirectly, are mostly usable when and where pressures are not too strong and can be diverted, stalled or anticipated. In sum, property rights constitute a powerful lever. Buying for conservation is up to competing with powerful economic adversaries of biodiversity – which may be what some of the critiques we discussed above reflect.

3.3.3. *A private involvement strongly intertwined with land, property and tax laws*

Like most market instruments (Kroeger and Casey, 2007), conservation easements are not possible without a specific, widely applied and powerful legal system (Gustanski, 2000). Whereas in some cases, land can be bought and conserved under unchanged legal conditions, in most cases, legal innovations are introduced to facilitate the use of conservation tools that rely on property rights. A first role they can play is removing blocks that would make buying for conservation difficult – for instance where laws put a strong property right or fiscal bonus on developing land. A second role is to make such transactions easier or cheaper. In the United States, two pieces of legislation have supported the emergence of easements such that it is now present in each of the 50 states (McLaughlin, 2005). The *Conservation Easement Act* facilitated reduced transaction costs for the establishment of easements by allowing state authorities to effectively waive certain property transaction fees. In addition, generous federal tax incentives encouraged landowners to compete for the easement purchases of states, creating real markets and lowering the cost of acquisition (McLaughlin, 2005). A third role is to facilitate management of land owned for conservation, by providing special provisions between owners and users or by a reduced fiscal pressure on land owned for conservation or bearing easements.

Even though this kind of ETB, being inherently private and monetary, might be considered as an alternative to public policies, the remarks above suggest otherwise. Even this seemingly "pure" market instrument relies on a strong coordination between market and public policy. At local and individual scale, buying for conservation is probably possible without strong cooperation from the administration. But when considering the prospect for its generalisation, the experience in the USA, where it is widely used, demonstrates a synergy between public organisation and private resources. To use only one example, buying land for conservation is operational only if the regime of rights allows owners to decide upon the (non-)use of their terrains, whereas in many cases it is not allowed to subtract surfaces from agricultural or forestry uses. Upscaling conservation on private grounds would require a co-adaptation of public and private initiatives; it would not mean "laissez-faire", but conversely a chain of policy and administrative decisions over many aspects: fiscal, land property and land-use regimes, jurisdictional, planning and sharing of responsibilities…

3.3.4. *Thoughtful planning is required*

While these tools possess, through their flexibility, the advantage of avoiding environmental conflicts, or potentially mitigating them, a disadvantage they suffer from is that they are inherently piecemeal in their approach. So an important issue raised about the use of conservation tools based on acquiring property rights is to what extent it allows deploying a coherent land conservation and management strategy at scales larger than individual estates. Buying property rights is inevitably subject to the contingencies of their availability for sale, which is usually unconnected with relevance for a coherent conservation program. Sometimes a willing buyer and a willing seller will not be found, at any price. Excepting the case of the purchase of very large individual estates that may have a stand-alone conservation value, programs of land buying or easements are best linked with planning measures that favour acquisitions converging towards a coherent set of conserved areas.

As in the Lake Yojoa case, in most projects involving the purchase of land or of easements, the local governments and communities are essential intermediaries between the donating land trust and the individual landowners, farmers, forest managers, etc. Their representatives are asked to organise participatory conservation planning, for example by defining targets and corridors where the purchase will be prioritised. In so doing, CE are closely related to land planning. They are based on preliminary zoning, which determines what lands are to be protected, developed, etc. When this zoning does not pre-exist, actors tend to trigger its elaboration.

First, this is another dimension involving some goodwill on the part of the authorities and stakeholders. The body owning the land or easements usually maintains a useful connection with planning authorities, and some standing in the planning process. As local land use plans evolve, the longstanding landowners are offered ample opportunities to contribute to the evolution of these programs, plans and policies. Owning land for conservation thus helps institutionalise biodiversity into the planning system.

Second, this reminds us that planning is also, and maybe particularly, a mechanism designed to manage land use. For example, the fiscal exemption regimes and the urban planning regulations are tools to decide over land use. Therefore, even a seemingly opportunistic private intervention, carried out in an emergency procedure, and where no ecological planning is in force, entails a planning regime that allows and facilitates it: the terrain must be free of restrictions, not pre-empted and other land uses such as agriculture or forestry must not be mandatory, etc. This is even more strikingly illustrated by the regime of development-permitting: where lands have been decided as non-constructible by law and regulation, this gives birth to interesting opportunities to buy less expensive land and preserve its biodiversity, even if subsequent acquisitions will happen (apparently) by chance. The synergy that exists here could be discussed in partly similar terms to the synergy between PES and legal provisions discussed in the previous chapter.

3.3.5. *Long-lasting management is quite a challenge, in contrast with the apparently instant character of the purchase itself*

With land ownership, which is a form of permanent control, local governance and environmental management become critical. It is one thing, for instance, to stop urban development on a coastal wetland by securing ownership, but the land then has to be managed in perpetuity. Conversely to commonplace critiques, keeping land for conservation is generally not excluding access and uses, but setting up a specific territorial management, with regulated access, uses, ecological objectives, maintenance, managed exploitation... To the owners of property rights on land thus befall the owner's responsibility for and often the cost of managing the land, and there is no exception for owners pursuing conservation. Once they have secured property, easements or concessions, the problem remains of defining and negotiating with stakeholders a management of the land and resources that is viable socially, economically, and in terms of the biodiversity goals of the initial purchase.

One might say that rather than the end of the worries, acquiring ownership is getting into (new) worries. In addition to technical and financial issues involved in such land management, the issue of local support or opposition can become very important both for negotiating management agreements initially, and for them to hold in time. Land trusts generally seek local management partners for the conservatories they establish. They have to negotiate with local stakeholders such as municipality councils, NGOs and communities as partners, seeking traditional local ecological knowledge, valuable services, financial and managerial input into management.

Long-lasting negotiation with stakeholders is a keystone of good management and obviously plays a central role in this ETB family. Skilful negotiation, before a purchase is made, is usually necessary to execute a successful transaction. Skilful negotiation during the purchase is critical to maintaining local support. Skilful negotiation after the successful purchase is also essential, if the final objective of the purchase is to be achieved. In other words, as an economic tool, the purchase of property rights creates conditions where management favourable to biodiversity can be put in place, whereas it would have been much less easy without an intervention in the market. But after the purchase, all the other aspects of negotiating land management for biodiversity come back to centre stage. All in all, "buying for conservation" ETBs are tools that create a situation where environmentalists obtain strong positions of negotiation in the management of the environment. As is the case with protected areas, most acquired territories are then subject to management plans, with steering committees, evaluation processes, and so on. Purchasing as a tool is but a moment in a wider sequence of actions. It neutralises the ruling of the market, somehow withdraws a given natural asset from the market rules, where and when administrative instruments are not suited or adequate. But after this subtraction is achieved, all common environmental management issues come back to the fore.

This demonstrates again that an ETB should not be considered as a solution *per se*, nor as an end in the process, and even less as a means to replace other resources such as negotiation, communication and lobbying, administrative intervention and legislative bargaining… Organisations that use purchasing strategies have to combine many capacities. Real estate management skills are imperative at one stage in the process, but those capacities are only components to be added to the equipment of the organisation. In strategic terms, for land trusts and all environmental actors that gravitate around them, it is crucial to acknowledge that common environmental management challenges will not be avoided by purchasing land or easements, and that articulating real estate skills with environmental management competences is essential. "Buying conservation" is only acquiring the right to engage in further environmental worries…

Being entitled to manage land and its uses over eternity, one has to expect that in the long run, most management parameters will eventually change. And permanence in a changing environment means periodical re-negotiations. Context, relations with partners in management, and the owning organisation itself, are not fixed. The continued protection of a particular piece of land may thus become impossible, impractical or undesirable. Or the management solutions that may allow for protection may change significantly. Environmental change is one obvious cause of such scenarios. The need for in-depth renegotiation of management agreement will therefore arise over time. For instance, the conditions that make extensive rangeland management viable now may well not be met in several decades. Also, if management has been largely delegated by the owner to managers, it may be reasonable to assume that with the passage of time, and with the managers investing money, knowledge and social mobilisation, the power standing relative to the owner may change, compared to what it was at the moment of the initial purchase when so much depended on the buyer, now owner.

Again, this paradoxical relation between real estate irreversibility and unstable management conditions is a specificity of this ETB. Other instruments do not hold this right to permanence, nor pretence at eternity. We showed here that the crux of acquisition strategies lies in goodwill, insofar as their intervention is possible only in a context where interests and willpowers are temporarily lined-up. However, the political legitimacy of acquisition is bound to be questioned in the long run, as well as its technical justification. Buying conservation is a one-shot intervention that subsequently leads to the need of a long-lasting negotiated management strategy. And at a given moment, because of environmental and economic changes, ownership will have to be asserted again as an essential resource in the powers balance in an environment that will have changed.

This necessary periodical re-negotiation should be kept in mind when assessing the opportunity for using the various variants of this ETB family. Land acquisition is ownership, whereas conservation easements mean joint ownership. If full property is more demanding in terms of financial means and institutional support, easements are also probably weaker when time comes to re-negotiate the political and technical legitimacy. Choices between different strategies will then have to take into consideration the stability of juridical and political regimes. This suggests, for instance, reserving soft and flexible instruments, such as easements, for contexts where regimes can be trusted in the long run, and conversely use sturdier tools, such as land acquisition, for contexts where instability can be feared. This could explain, besides, why CEs are mostly used in a country with comparatively very stable property institutions, like the US.

3.4. "Purchasing biodiversity" creates strong, skilled and dexterous environmental actors

ETBs do not work by themselves. Voluntary instruments like the ones discussed in this chapter require a human ecology of their own. They need certain actors, which in turn spur the development of the instruments.

First, these ETBs tend to strengthen actors specialised in conservation, by giving them the opportunity of developing and supporting large land-owning organisations devoted to conservation. Their presence "around the table" where land and resource use are discussed, or on the side of conservation advocates, of organisations with substantial means, is an asset for advocates of biodiversity. This is an opportunity to strengthen the cause of biodiversity conservation. In terms of the position of land and land rights acquisition-based tools in the overall strategic action for biodiversity, this is an important consideration.

Second, it is important to stress that public support is needed on an ongoing basis. It is required for the initial passage of regulatory structures to facilitate and encourage property-based conservation. It remains critical as these regulations are transposed into local contexts and implemented at regional and local levels. It then needs to be maintained as perverse incentives are identified and rectified, and the regulatory regime is bedded down. It must also be politically defended against critics. Buying land on behalf of the public, especially when it is opened to the public, can also have long-lasting effects in terms of public support to conservation.

Last, private conservation efforts require brokers, both within and outside the trusts and NGOs that pursue and promote these transactions. A community develops which may enhance the ability of parties to find one another, and to negotiate mutually beneficial agreements. For example in Florida, pricing is generally based on the preparation of appraisal reports from certified third party experts known as 'appraisers'. The development of such brokering activities may also have overall positive effects in terms of the presence of conservation-expert professionals in the arenas of land-use negotiation and planning more generally (Boyd *et al.,* 1999).

In sum, when analysing the subject of their actors, these ETBs appear as tools that reinforce their own users, and that can even create actors. The aforementioned example of TNC, and the comments it attracts regarding its economic power, are remarkable in that respect. This ETB is able to render its users more professional, wealthier and more powerful, and more connected to other actors on the ground.

Land ownership, in the long run, is a source of power. And being owner ad vitam aeternam, with irreversibility, produces a possibility to progressively accumulate power and strength: an organisation can rely upon its historic assets, while keeping on accumulating. Whereas PES and, we will see, other ETBs, are mostly *ad hoc* constructions, "buying for conservation" tools are the most susceptible to institute an environmental "sector", as did, in their time and through different ways, the regulations that created and organised natural reserves. It does not grant actors the same administrative power, but it grants them a patrimonial wealth, a need for expertise and means to acquire and exercise it, and the power of the owner.

Conclusion

An *a priori* and superficial analysis of these ETBs would suggest they are purely private and unilateral tools, and that they are more independent from the institutional and administrative system that is typical of so many environmental management instruments. If, indeed, acquiring land or land-based rights on the market and reserving them for conservation are market instruments with a vengeance, this chapter has tried to clarify and discuss their strong interaction with their social and environmental contexts.

The analysis of these instruments "in use" has allowed us to suggest fairly different perspectives than those that derive from critiques based on the principle of purchasing biodiversity conservation rights. Our short review of these critiques has revealed that they are not mostly questioning the efficiency of these ETBs. They do not much discuss their merits for achieving ecological objectives, for instance. They rather object to the social and economic consequences of allowing a strong environmental intervention, of changing the balance of powers, and of obtaining environmental results that would interfere with other conceptions of development and social objectives.

Our analysis "in use" brings essentially three different points of view. (1) Far from being a blind takeover strategy, intervention on land property for conservation goals depends on a conjunction of good wills, and needs interacting with land planning; it requires concrete adaptations of the prevailing fiscal and juridical rules, and an enduring dialogue with other powers and authorities. (2) This ETB brings a potential to create and/or reinforce powerful environmental actors, and moreover to foster the emergence of an environmental sector. Besides, this sector is potentially equipped to step in against strong economic drivers such as urbanisation, or conversely to secure conservation in areas where pressures are yet very low. (3) Acting as an owner in the long run means entering into a long process of negotiated environmental management, and preparing to recurrently re-negotiate the balance of powers and rights.

Subsequently, this analysis now points to the need to understand how this lining-up of good wills should be obtained, and how good wills will materialise into fiscal and juridical provisions. It raises questions about how to manage this ability to create and reinforce environmental actors and sectors: how should they relate to other actors, how should alliances be made, how should complementarity and opposition be managed with other drivers and other instruments? Lastly, it suggests a better understanding of how to deal with justification, how to negotiate regimes and management and how to live, in the specific situation created by these ETBs, with changing political, environmental and economic contexts. Those issues will be dealt with in chapters 5 to 10, while the next chapter is turning to the fourth and last family of ETBs in our list, compensation and exchangeable rights.

4. Trading conservation & biodiversity: a heavily administered market

Introduction

Apart from subsidising a change in producers' behaviours as in PES, or treating biodiversity as an asset through the purchase of land or land uses, another way to address biodiversity issues by way of an economic instrument is by imposing access rights on biodiversity units, and allowing agents to destroy some of them in exchange for the creation of others, and/or to exchange these rights. In other words, it means artificially creating a commodity equivalent to biodiversity, which then can be managed just as for privately-held resources. This approach is more or less, with variants, at the heart of instruments such as tradable permits, exchangeable quotas and compensation offsets.

In other words, some see the possibility of such instruments with a hope that biodiversity will be managed by the market and thus replace a rigid, heavy and costly "command-and-control" administrative management. Both the literature and our interviewees often refer to the precedent of carbon credits to advocate biodiversity banking. Even though carbon credits are not yet fully effective, they have concretely given birth to exchange markets, brokers, price curbs and speculations, and this example probably acts as an ideal benchmark for many. Why not imagine biodiversity being managed as it is conceived for other intangible goods such as carbon quotas, mobile telephone frequencies, etc.? Will this not, at last, simplify management, spare heavy administrative burden, avoid inquisitorial regulations, etc.?

This chapter will be organised as the previous one. (1) We will briefly define and illustrate this family of tools. (2) We will then probe into the foundations of the tool and examine the arguments put forward by its promoters and critiques respectively. (3) Finally, we will examine more in-depth the challenges raised by the scope and the organisational functioning of such instruments in actual use.

4.1. The "trading conservation" family of ETBs

Offsets (or compensation) consist in systems whereby an operator receives the right to destroy a unit of biodiversity in exchange for restoring or for conserving an equivalent unit of biodiversity somewhere else. Biodiversity banking adds to the same mechanism the possibility of trading: an operator can receive the right to destroy a unit in exchange for buying a credit from another operator who has restored or conserved it somewhere else.

4.1.1. *Examples*

To complete this presentation, let us introduce two examples of offsetting and biodiversity banking respectively.

The first, the Ambatovy project, is located in Madagascar. A number of firms and NGOs have been collaborating in recent years to develop rigorous principles and practices for biodiversity offsetting. One such collaboration is the Business and Biodiversity Offsets Program (BBOP).

Under this program, a large cobalt and nickel mine in Madagascar, partly financed by the European Investment Bank (EIB) is seeking to achieve a 'net positive' outcome for biodiversity, through applying the mitigation hierarchy and an offset program. The measures being implemented include an offset site containing a core conservation area designed to compensate for the residual impact of the mining project, which cannot be mitigated. To ensure landscape level connectivity, a residual forest corridor linking forests surrounding the mine to a national park has been set aside, and targeted reforestation of additional corridors is being undertaken in conjunction with local government and NGO parties to re-establish connectivity in more effective scales than exists at present in the region. These activities, along with progressive rehabilitation at the mine site to produce a rich forest ecosystem with reinstated biodiversity values should together ensure that the net impact of the Ambatovy project is positive for biodiversity in Madagascar.

Our second example will be the New South Wales (Australia) biobanking scheme.

This scheme is established under the NSW Threatened Species Conservation Act and provides a specific pathway for a developer to deal with its impacts on biodiversity. Developers can prepare biodiversity compensation projects and then run through an assessment of how they will positively affect biodiversity values, and receive a score in the form of a credit value, which can be sold afterwards. Credits are generated through the imposition of a conservation easement, the enacting of a restoration or management plan, and calculation of the cost in perpetuity of good environmental management of the land, which sets the floor price of any trade. Credits are in two forms: ecosystem credits (for a habitat type) and species-specific credits. A project will be ascribed a score for both types of credit.

When considering an authorisation implying a depletion of biodiversity, the consent authority will require the developer to purchase an amount of credits equivalent to what it intends to destroy.

Conservation priorities in the form of scarce species or ecosystems are given preference through the operation of the biobanking market mechanism. For example, if a development is occurring in the relatively abundant sandstone transition forest, credits may be purchased for analogous ecosystem protection anywhere up or down the NSW coast, because the same assemblage of species is present in a wide variety of coastal locations. But if it's Cumberland Plain Woodland, the scarcest habitat type in Sydney, credits will only be able to be generated in the flatlands of Western Sydney. This scarcity increases its value. Therefore the market is supposed to drive developers away from it, and to encourage those that have high quality stands of such habitats to conserve them, as they can sell those credits for significant amounts of money.

4.2. Foundations, positive expectations and critiques

In the field of ETBs, biodiversity banking is probably the issue that is currently attracting most attention and generating the most heated debates. A large part of the controversy bears on principles. We shall not try to go to the bottom of such principle issues here (readers particularly interested in that issue may find useful equipment to address it in chapter 8). But recalling some salient points of the controversy and clarifying some essential issues is necessary before we turn to issues in practical use. Let us examine the foundations of offsets and biodiversity banking, some of the arguments of its supporters, and then of its critics.

4.2.1. *Roots in three distinct concepts and practical precedents*

Tools relying on biodiversity offsets and biodiversity banking have their origin in three distinct sources of inspiration and founding concepts: environmental impact assessment, tradable quotas, and "no net loss" habitat conservation policies.

(a) *Offsets* are a logical development of *environmental impact assessments* (EIA) of development projects, a major tool in limiting the impacts of development on biodiversity, that has been put in place gradually all over the world in the last four decades. The rationale of EIA goes through several steps. The first is to identify impacts of a development project and its variants. If there are significant impacts, a second step is to see whether the project may be reconsidered, or another variant chosen, for instance through choosing another location, or another basic project design. In this way, some of the identified impacts may be avoided. A third step is to mitigate impacts, for instance through the reduction of the surface area subject to impacts through appropriate design and implementation of the project, or through biodiversity rescue and relocation programs during forest clearance, or through biodiversity friendly management of less intensively used areas in the projects' perimeter. The following logical step is to compensate for remaining impacts that could neither be avoided, nor be mitigated. This is the basic principle of biodiversity offsets: to obtain a permit for a project that will degrade or destroy some species or habitat, or impede some ecosystem function, the authors of the project will have to compensate by protecting, restoring or recreating equivalent elements of biodiversity elsewhere. As a tool, offset does precisely what EIA and impact mitigation cannot do: ensure that a loss of biodiversity somewhere is compensated by a gain elsewhere. If the alternative is a loss without compensation, then the tool clearly has a potential to be useful.

(b) *Biodiversity banking* consists in trading offsets. This is where offsets, which are not *per se* an economic tool, become one. Additionally to the principle of offsets, the concept has taken inspiration from *environmental economics,* and the gains available through trade (ten Kate *et al.,* 2004). The earliest iterations of these tools in the field of living resources are single species tradable quotas (Individual Transferable Quotas, ITQs) used in fisheries management. ITQs were introduced in the 1980s in Iceland and New Zealand as an additional management tool (focussing on output) to complement existing input controls and conservation restrictions in fishery regimes (Grafton and McIlgorm, 2009). These quotas seek to capture efficiency and transparency gains from trade between fishermen to achieve both more certain biomass extraction and enforcement cost savings. The additional contribution expected from biodiversity banking (compared with direct offsets) is to allow obtaining compensation more cheaply, both by being able to choose amongst alternative offset sites competing on a market, and by reducing the high transaction costs involved in organising compensation on a case by case basis, through the existence of an offer of restored or conserved biodiversity ready to be traded in exchange of negative impacts that need compensation. Banking thus appears as going one step further than offsetting towards economic instrumentation. It disconnects the consumption of biodiversity and its re-creation, allows sparing, storing, accumulating and exchanging biodiversity units.

In that perspective, in addition to the above trading schemes, some financial institutions are now seeking to push further the concept of biodiversity banking by cutting clear of offsets and developing *economic biodiversity credits* (EBCs). ECBs seek to compensate the economic value of biodiversity destroyed in development, rather than the ecological assemblages (EFTEC and IEEP 2010). This tool would seek to quantify the total value of ecosystem services, in addition to existence values, and then spend an equivalent amount of money on ecological restoration. Such a tool would break the nexus between biodiversity and its financial value. Rather than seeking some form of like-for-like compensation or preservation, EBCs would seek a like-for-like financial equivalence, in the economic cost to society of a particular instance of biodiversity destruction. Such schemes have not been put to test yet, however, and existing biodiversity banking experiments consist in trading offsets, guided by ecological equivalences.

(c) The final and third pillar of offsets and biodiversity banking is the concept of "*no net loss*" or, more generally speaking, of putting *a firm regulatory* cap on the overall acceptable level of biodiversity destruction. The wetland and endangered species regimes in the United States, put in place in the late 1980s and in the 1990s, set the most influential precedent. Here the state declares a policy of 'no net loss' and requires development activity which causes wetland destruction to remediate an equivalent area of wetland in the same catchment area. A similar system operates for development adversely impacting on endangered species. An endangered species credit, showing the restoration of the species, must be delivered to offset any loss of the species due to proposed development. Over time, more sophisticated versions of biodiversity and habitat banking have been implemented in jurisdictions such as Australia. The New South Wales 'biobanking' scheme for example, requires any development to assess both the species and the habitats it will impact. An equivalent, protected habitat and set of species assemblages must then be conserved in perpetuity, as a consent condition for development.

It may be noted that whereas "no net loss" is the simplest concept on which to base offsets and trades, it is not the only one. Biodiversity offsetting, as compensation for destruction, can offer also "trade-up", by allowing the creation of higher biodiversity spaces in lieu of degraded lower areas (ten Kate, Bishop *et al.* 2004).

4.2.2. *Expected benefits from biodiversity banking*

Not surprisingly, biodiversity banking receives support based on each of its three roots.

From an EIA perspective, offsets are a good thing, because it s possible to be more demanding of operators with biodiversity-damaging projects. The obligation to compensate and the increased feasibility (as operators can easily find on the biodiversity banking market compensation that they can simply purchase) provide the regulators that use EIA with a stronger negotiation position in the face of biodiversity-damaging project operators.

From an environmental economics perspective, the tool is praised, in principle and in theory, in many textbooks and economic theory classics (Pearce and Turner, 1990; Tietenberg, 1990; Cornes and Sandler, 1999; Freeman III 2003; Pearce *et al.,* 2007). Indeed, such instruments are supposed to possess many of the virtues of a functional market: because rights have to be bought, and because quantities and prices will adjust to the willingness to pay of economic agents, the regulator does not need to know much of the economic conditions of agents, and the distribution of efforts to preserve the environment will be adjusted swiftly according to the market forces. Firms with low abatement costs will be able to achieve additional abatement beyond their own needs, and then sell this extra abatement to firms facing higher internal abatement costs, thus providing the "gains for trade" (Whitten *et al.,* 2007). The desired abatement is thus deemed to be achieved more cheaply for all firms (and hence society) than if a more direct regulatory requirement were imposed on firms, for example if each polluter had to reduce its pollution by the same proportion.

Finally, the "no net loss" foundation of biodiversity banking attracts broad-based support from constituencies that express biodiversity concerns. The clarity and ambition of the idea of putting an end to biodiversity loss is a powerful attractor for politics and policy.

4.2.3. *Vocal and pointed critiques*

But offsets and biodiversity banking are also the target of vocal and pointed critiques.

Perhaps the most striking critique, offered by both philosophers (O'Neill, 2011) and ecologists is that the fungibility of habitats (or species) required for any operational market is highly questionable both in principle and in practice. Fungibility is a form of equivalence, it is saying that place X is in some way equivalent to place Y. From a philosophical perspective, the human non-economic value pertaining to one specific place can never truly be equivalent to another place. In practical ecological terms, it is extremely difficult to recreate an identical assemblage of organisms, except within fairly loose guidelines (Predal, 2010). This is a particularly strong critique since it attacks the very foundation of offsets and biodiversity banking: the fact that some biodiversity lost somewhere can be compensated by somewhat equivalent biodiversity restored somewhere else. This is one point on which the difference between biodiversity and climate is acute. In terms of greenhouse effect, one ton of CO_2 emitted anywhere is comparable: the carbon market is based on a high degree of fungibility. By contrast, biodiversity, as the word itself proclaims, is fundamentally predicated on heterogeneity, variability, and often, uniqueness. Even if some trade-offs can make a lot of sense, fungibility can never be taken for granted here.

A second widely shared critique of all schemes based on trading compensations in exchange for the permitting of activity with negative impacts (both biodiversity and pollution permit trading schemes) is that they can be understood as providing a "licence to trash". A first version of this criticism is moral, and rejects the premise of "no loss", or "net gain", focusing instead on the potential for the wealthiest firms to effectively buy a right to pollute or destroy: private wealth would give a privileged access to a global heritage, and a licence to destroy it. A second, strategic, version of this objection is scepticism regarding the notion of "unavoidable" impact. The reasoning is that if impacts can be compensated, then the permitting authority will be tempted to give permits more easily, it entails the risk of "opening the door" wider than without the compensation system. As a result, more primary impacts would be authorised and, compensation being necessarily imperfect and incomplete, the gain it provides would not equal the loss of increased primary impacts. In other words, if trading compensations is a way to make it easier to meet a fixed limit (on pollution, on the level of biodiversity that can be destroyed), it can easily shift and make implementation of the limit more lax.

A third frequent critique is that trading brings privatisation in its wake. This critique holds that whereas it may be fine to set a clear (*e.g.* "no net loss") ceiling to biodiversity loss, managing this limit through the trading of rights lead to the disowning of rights (and the oligopolisation of profits) that tends to accrue from the trading itself. One may even write that here, ecological limits, or limits set to the degradation of ecosystems, become a market good that is traded. Depending on the way this trade is organised, it does not enrich (or empower) the same sort of operators. In the eyes of the critics, the actors that may benefit most from biodiversity banking (banks, large scale brokers, large firms in the sectors involved) can, through the privatisation of land, habitats or resources formerly either held in small holdings (small-scale fishing or grazing rights), by communities, or publicly (treating the management of water in ecosystems as a public good, for instance), actually dispossess current users of amenities or resources.

The experience of ITQs in the field of single species fisheries management has demonstrated that this is a very real risk indeed. The implementation of ITQs, for instance in Iceland, has triggered a "rationalisation" of the structure of the fishing industry to fewer, more centralised corporate entities, and also a sharp reduction of individual owner-operators. Ex-post analysis of national experiences in that field (for instance in Iceland) have shown that the complexity of trade has acted as a catalyst for rationalisation of the industry, leaving only a very limited number of players with considerable expertise and resources to be able to compete for appropriation of entitlements and may have many negative social effects.

4.3. Issues in use

The main issues raised by the critics of offset and biodiversity banking point to several issues that can be very challenging in the actual design and implementation of offset and biodiversity-banking schemes. They occur at the various successive stages of the operating chain of the tool: (a) forbid any avoidable negative impact of projects on biodiversity; to manage remaining unavoidable impacts (b) set a cap on biodiversity loss and demand compensation, (c) establish equivalences and encourage trade. These three stages are vulnerable respectively to (a) the "licence to trash" effect, (b) problems with limits set to biodiversity loss, (c) issues of fungibility. Let us examine successively how these challenges play out in practice for operators of offset and biodiversity banking schemes who really want to get tangible benefits for biodiversity from the tool.

4.3.1. *Offsetting is not licencing to trash* per se, *all depends on the chain of permitting*

Compensation is the last link in the "avoid impact", "mitigate what you can't avoid", "compensate what you can't mitigate" chain of reasoning and regulatory obligation. The temptation might be very strong – and quite rational – to re-examine avoidance and mitigation efforts in view of the possibilities for compensation. If compensation is relatively easy, then relaxing efforts to avoid and mitigate negative impacts is the logical step for the developer. For the authorities, if their aim is to foster development, trading ETBs may lead to becoming more flexible in their permitting.

For radical critiques, as we saw, the existence of such a shift is in itself an argument against the very principle of offset and biodiversity-banking. But the shift towards obtaining more easily a "license to trash" may also be incorporated in a more complex balance of gains and losses for biodiversity. In principle, it may very well be that in some cases, if one accepts a small shift in avoiding (put simply, if one gives more permits for building in a given area in exchange for an ambitious compensation program), the net balance for biodiversity may well turn out to be positive. More generally, the flexibility offered by compensation is to be examined relatively to the current situation of biodiversity protection decisions. For example, in regions where limited impact mitigation is the only required condition to authorise a development, and when this is due to a difficulty experienced by authorities in refusing projects, then one can't say that offsetting would "open the door", because the door is already open. Rather, offsetting could help at least partly close the door.

This certainly does not put compensation in the clear as if it automatically provided a satisfactory solution. What it shows is that evaluation of biodiversity outcomes should be directed at the entire chain of permitting, including all stages: avoidance and mitigation in a system without compensation, avoidance, mitigation and compensation in a system with compensation. Because compensation inevitably leads to reconsidering levels of efforts for avoidance and mitigation, its biodiversity benefits cannot be assessed independently from the entire permitting procedure.

What is decisive in practice is the existence of collective action to ensure that permitting is done under conditions that guard as closely as possible the door to "trashing" and that the entire permitting chain is dealt with in a way that delivers the biodiversity goals that are sought, whether they be "no net loss", "net benefit", or some form of "some loss in exchange of stronger conservation action somewhere".

4.3.2. *Clarity and firmness of the cap are of the essence*

The practical design and evaluation of offset and biodiversity banking is highly dependent on the clarity and firmness of the cap set to biodiversity loss. In practice and in debates about the two, we find that much ambiguity in this area sometimes compromises the tool in terms of its usefulness for biodiversity. Let us re-examine clarity of the cap, then its firmness.

In terms of the caps they are based on, current biodiversity trading schemes appear to fall into one of three categories:

1. *Net gain schemes.* The US endangered species trading schemes are good examples of this policy goal. As an increase in the relevant population of endangered species is the goal of the legislation, brokers will seek out breeders and restorers of endangered species, to fund, grow and protect new populations of these species to more than compensate for species losses which might arise due to development activity, and to be achieved prior to the granting of consent for the project.
2. *No net-loss schemes.* No net-loss standards have been imposed by US wetland banking schemes, which require the restoration of a similar area of wetlands to be completed before existing wetlands may be destroyed. The goal of no net-loss is imposed by legislation with trading schemes within watershed basins arising to link private landholders who are restoring wetlands with developers wishing to denude existing wetlands.
3. *Some biodiversity loss in exchange for some conservation somewhere else.* In some schemes, unavoidable impacts can be compensated by improving the protection status of some other area (by giving it protected area status, by making it a private nature reserve). In that case, there is some loss of biodiversity, but there is better prevention of further loss. Such arrangements compensate from a loss of biodiversity by a stronger conservation shield on similar elements of biodiversity somewhere else. It is important to note that such arrangements do not guarantee no net loss (and even less net gain): the biodiversity that existed in the newly protected area already existed, while the one that is going to be destroyed will effectively be lost.

In each of these three types of caps, the meaning of offset and banking is very different. And yet, many schemes are unclear in terms of what kind of cap exactly serves as their baseline for design and assessment of effectiveness. Clarifying the type of cap is a necessary step forward to improve use of the tool in many situations.

But being clear is not enough: effectiveness of the tool also crucially depends on how firm the cap is. Actual behaviour of project operators – and thus actual impact on biodiversity – depends not on posted principles, but on actual pressure or obligations. In biodiversity banking developers purchase offset credits because it is the only, or the best option to get a permit. If, however, there are ways to avoid or minimise this obligation, the value of the credits can only flounder and the whole system is likely to fail. Absence, in actual practice, of firm caps to biodiversity depletion may then be one reason why, contrary to their salience in the biodiversity and economic literature, and conversely to PES, biodiversity trading schemes are scarce in the world, except maybe in the USA and Australia.

Indeed, the firmness of caps is not only a matter of regulations. It is very much an issue of implementation. EFTEC and IEEP (2010) examined the regulatory conditions for biodiversity offsetting to be used in Europe. They showed that the legislative apparatus, in the EU and in most member States, does not need any further development to allow for biodiversity offsets to be put in place. Only two ingredients are missing: (1) the clear affirmation by governments that the intent of the regulation is actually to reach a firm cap (*e.g.* no net loss) to biodiversity destruction and (2) their commitment to implementation of regulation to a level of effectiveness that makes these caps a tangible reality, solid enough to found a market on it.
This suggests that, except in the case of the no-net-loss policy, what is missing most often for an offsetting or banking system to take place is a political and thus societal agreement on imposing a halt to the continuous loss of habitats. Legislative means have been adopted, but they are to be handled within an ocean of legislations, and their mobilisation in specific areas is missing.

However, economic instruments such as biodiversity banking are sometimes advocated as proposals that may facilitate the decision-making that may lead to adopting firm caps. The rationale here is that the decision of establishing firm caps to biodiversity loss is currently stalled because of dreading high costs for the administration and the economy, expecting inefficient implementation, and because of the impossibility to gather all technical and economic information needed. Designing, testing and proposing economic instruments such as biodiversity trading is then supposed to suppress those blockages and help firm caps to be set and implemented. In the case of biodiversity trading, considering the paucity of reported examples, such effects are not obvious yet. This suggests either that, conversely to these assumptions, achieving a more ambitious biodiversity conservation policy is not precluded mostly by efficiency criteria. Or it can mean that such criteria are indeed crucial, but trading ETBs are not providing sufficient guarantees yet.

In sum, this family of ETBs is one more example of situations where the availability of technically feasible and economically efficient instruments does not, at present, appear to be the decisive condition for a turnaround in our management of biodiversity, even if it can contribute to it. Where there is no sufficient political and societal commitment to halting biodiversity loss, issues with the tool are largely academic (in terms of biodiversity impacts). Where the commitments are strong enough, then the quality of use (design and implementation of the tool) becomes a central stake.

4.3.3. *Establishing equivalence is a difficult and contested task*

How equivalence for compensation is established is then a key issue in offset and banking schemes.

In terms of ecological equivalence, a tension is inevitable between strict guidelines, which segment a landmass into more habitats and ensure that greater distinctiveness is preserved, as against more flexible guidelines, which will facilitate a greater potential set of trades for any one development but may less closely approximate ecological equivalency. This may explain why, as is the case for valuation for economists, rationalising instruments are more and more developed and discussed so as to offer technical solutions – dynamic and ecologically sophisticated GIS tools that can map out existing functional connectivity for representative species of various habitat types, site selection tools to guide the location of offsets... (EFTEC and IEEP, 2010) – when this kind of ETB comes into force. They are means to address the ecological fungibility issues, but it is doubtful that the solution will lie mostly in more sophisticated tools to characterise the biodiversity that is lost and gained. Continually refining the criteria to be taken in charge by trading schemes would mean mechanically creating a growing number of different "ecological goods" to be considered as equivalents. As economics puts it, it means increasing the number of different "markets": a market for this taxon, this specific combination of species, this landscape, etc. Eventually, equivalences and possibilities to compensate would inevitably shrink. However, the existence of a market for a definite good means there is a reasonable number of units of this good to be traded. Trading biodiversity therefore involves exchanging ecological goods that are not exactly equivalent; it means exchanging some kind of biodiversity loss with another kind of biodiversity gain. It is in managing these necessary non-equivalencies that trading schemes will or will not produce a satisfactory ecological result.

In other words, hardly any compensation, in any field, ever really replaces identically what has been lost – think for instance of personal damage compensation for bodily injuries. This does not invalidate the principle of compensation: no compensation is not a better alternative to being compensated with something somewhat different than what is lost. The equivalence cannot be established in an absolute way. Judgement is fundamental, be it in court, in political arbitrage or in negotiation between parties. In terms of biodiversity, the essence of offset-bases schemes is that what is obtained in terms of compensation be felt by those stakeholders most engaged in biodiversity conservation and restoration to re-establish or improve the biodiversity situation overall.

4.3.4. *Acceptability: offsetting does not suppress having to deal with local changes and inequalities*

This criterion, however, addresses only biodiversity and leaves aside other socially important limits of fungibility. Even if there is adequate biodiversity compensation, value equivalence can remain problematic in terms of amenities to the people who value specific places. In practice, being based on compensation for damage in place A by conservation in place B, offset and banking schemes raise issues of acceptability both in place A and B. For people in place A, the permitting may well mean a loss of amenities and other ecosystem services that may not be conserved for them by compensation in place B. The last meadow or forest patch near their home is in no way replaced by a meadow or forest patch – even fuller of flower, birds and crickets – further from home. As for people in place B, compensation effectively means that there will be a conservation scheme in perpetuity, raising the same possible concerns as any other such tool, whether it be a protected area, or land acquired for conservation: they may feel that this is short-selling their own possibilities for development for the benefit of the people in place A.

What appears here is that, beyond being a tool to exchange piecemeal development against conservation, trading on biodiversity is part of overall planning and development policy. The political acceptability of banking schemes will in part be determined by their interplay with other planning and conservation tools and priorities.

Overall, non-ecological priorities for environmental policy, such as the preservation of open space and the enabling of a connection to the natural world, are not well suited to being met through this policy tool. And since they are important elements in the acceptability of decisions affecting the environment, they will have to be dealt with on their own terms, in parallel with offsets and biodiversity banking schemes.

4.3.5. *Permanence is an issue as for other ETBs*

To this list of practical issues that closely follows the contributions of critics of the principle of offset and biodiversity banking, there is a need to add one issue raised by examination of the practical schemes currently being experimented: the integrity of banking schemes in the long run. This integrity is critical to the ecological outcomes: if one has traded permanent development (housing for instance) in exchange for non-permanent restoration or conservation, biodiversity loss has been postponed, not halted. With trade values based on the costs in perpetuity of conserving and managing lands, a rigorous enforcement of these legal undertakings must be present. But the issue is not just legal. We have met very much the same issues as in tools based on the purchase of property rights for conservation. The perpetuity that is inherent in both families of tools raises issues in the long-term management of land, of habitats, of ecological systems, and in the perpetuity of the holders of the rights. Biodiversity banking institutions that are experimenting with biodiversity banking schemes today in France sell credits for restoration and management of habitat for thirty years. They indicate clearly that subsequently, arrangements will have to be re-examined. Here, the practical limits of "perpetuity" are quite straightforward. But even when limits are not clearly expressed, they do exist, for instance in the possibility that organisations holding the rights and obligations may not continue to be in existence with unchanged mandates forever.

This may not be a major practical issue at the stage of pilot experiments. But as soon as the scale of application increases, it is a major aspect of tools based on compensations that are expected to be perpetual. In terms of ecosystem functioning, of biodiversity, but also in terms of large-scale public policy and legal systems, thirty years is not a very long time indeed.

4.4. A change of scale in biodiversity administration efforts

All the implementation difficulties we just reviewed do not suppress the potential usefulness of offset and banking schemes. What these can provide – balancing inevitable (or socially desirable) damage to biodiversity with biodiversity gains – is a necessary component of any "no net loss" or "net gains" goals for biodiversity (bar the possibility of a total status quo in land use). As we just discussed, these tools can function properly only if certain conditions are met: a firm regulatory cap, complex rules of equivalence that rely on much scientific and expert input, insertion in the overall chain of authorisation, solving permanence and local acceptability issues. Offset and biodiversity banking, if they are to be used with good effect on biodiversity require that we move to new levels of standards and efforts in the administration of biodiversity.

4.4.1. *A whole new level of administration load*

First, the system requires massive administrative involvement of public authorities. To run an offset system, the administration needs to be "independent, expert, adequately staffed and financed, and open to scrutiny and auditing" (EFTEC and IEEP, 2010, p. 142). This vision of the system leads, it seems inevitably, to propose new governmental bodies to take charge of those heavy burdens, given the situation of current administrations. Hill (2009) recommends creating an agency to take charge of the regulation of planning authorities and ensure that damages are monitored and constraints are correctly enforced (note that this new public body would regulate first other public bodies, those in charge of planning).

Bottlenecks for implementation are thus in public capacities. Even when mentioning the readiness of other stakeholders (NGOs) to take charge of some aspects of the system, once again, that this would mean they are remunerated, and this would be funded through an administrative charge to developers, included in the fee... As it appeared early in environmental economics history, and in experiences in tradable quotas, these are tools that require a very heavy administrative load, involving lawyers, scientists and ecological engineers, administrative personnel, facilitators and participation specialists, etc. The possibility for trading biodiversity provides no alternative to the need for high levels of scientific information, for a high level of planning controls, or high levels of legal obligations. On the contrary it requires all those resources and conditions. And only if these are provided can compensation then provide additional possibilities in the search of better biodiversity outcomes. This is again a very different picture from depicting a scheme that would operate with downscaled administrative intervention.

4.4.2. *A high ambition, but for a limited set of activity sectors*

A striking feature of the literature and expertise on offset and biodiversity banking schemes is that the field of application is restricted to a short list of development sectors: urban and industrial development, infrastructure (especially transport) and mining. These sectors share two characteristics.

First, they create much economic added value, compared to the area surface that they consume. When developing a housing scheme, building a plant, or digging a mine, the range of habitat destroyed is moderate with regard to the funds expended – if one compares with other sectors such as agriculture or forestry. As a result, there may be significant resources to provide for compensation and an amount of necessary compensation that remains within workable limits. A housing scheme filling two hectares of wetlands can generate much added value and realistically fund the restoration of equivalent or superior wetland values somewhere else.

Second, these are all sectors that already have a long history of EIA procedures being the main environmental management tool that they use, especially for biodiversity issues. And as one interviewee explained, since offset is just the next logical step to extending that procedure, it is easy for those operators who are already operating within that procedure to adopt it too. Many of the needed elements are already in place: permitting rules and relations with administrations implementing it, expertise for the assessment of biodiversity and the impacts of development, negotiation on avoidance and mitigation of impacts. Adding further rules and negotiations for compensation is indeed only one further step to add along the same pathway. One may add that compensation often consists in buying a site, doing engineering work on it and then managing it – with the nuance that this is ecological engineering. This is very much the sort of professional activity that developers in the sectors involved are familiar with.

The same factors that facilitate adoption of biodiversity trading for these sectors seriously limit its potential for use by others. Consider for instance the intensification of farming, which is the main cause of biodiversity loss in Europe. This mostly occurs on very large areas, generating only limited economic added value relative to surface (compared with housing or transport infrastructures for instance). Agricultural intensification relies on the conversion of land under light farming use – often with high biodiversity – into land under heavy agriculture use – usually with much less biodiversity. Offset would require that equivalent amounts of land be returned to light agricultural use… This runs contrary to the whole large-scale trend of agricultural intensification and the policies that promote it and is just not feasible in the EU context.

The scope of offsets and banking is thus limited to those sectors where a balance between biodiversity destruction and biodiversity restoration is achievable. This is not the case in those fields of activity that involve the transformation of large areas from high biodiversity to lower biodiversity use (such as agricultural intensification, palm oil plantation or exploitation of old forests), where this is just not possible. In sectors like agriculture and forestry, the scope of these tools is limited to those projects where there is high added value (for instance, greenhouses for vegetables, flowers or fruit) or to those areas where agriculture expansion is halted (for instance, mature agricultures in some areas of the EU where biodiversity impacts in intensified lands can be offset with improvement in the conservation and management of more marginal farmland).

This difficulty in the overall biodiversity and economic balance of some sectors is compounded by the fact that many industries are not, or not yet, under regimes of EIA and permitting prior to implementing the technical projects of their choice, in which case the use of tools relying on compensation of impact is much more difficult.

Practitioners we have interviewed concur in defining a relevant but limited scope. They suggest that offsets are meant to address a kind of pressure that is not major in the problem of biodiversity. The literature suggests likewise: "*Although the impacts of most appropriately located and mitigated infrastructure projects in the EU are relatively low compared to other pressures, many cause significant biodiversity losses.*" (EFTEC and IEEP, 2010, p.19). By their cumulative numbers, infrastructure projects are still an important pressure to address. Thus, the Australian NSW contrasts an old approach, where benefits from development were always seen as outweighing losses of biodiversity, with a new approach, based on the "tyranny of small decisions" vision, where it is advocated that those small decisions lead to "a downward spiral of continuing incremental biodiversity loss" (NSW Department of Environment and Conservation, 2005).

That is, offsets and banking are meant to be localised and selective, as opposed to widespread and diffuse instruments that are, for any reason regulated under other instruments (CAP, etc.).

Heavy requirements in terms of administrative enforcement and workload also limit offsets and banking to areas where a rather strict protection is enforced, and where an effective legislative and regulatory system is in place. In such regions, Cannon and Brown (2008) consider that banking is particularly suited for mitigating a certain category of problems: the accumulation of small projects, each having detrimental effects but which would otherwise probably not receive sufficient attention to provide environmental protection.

In sum, offsetting and banking may be seen as relatively complementary to other ETBs such as PES. PES are essentially equipped to deal with intermediary situations where beneficiaries or "victims" are in a position to pay off producers so that they modify harmful practices or sustain beneficial ones. But when the profits from biodiversity depletion are high, and so, subsequently, are the opportunity costs from renouncing it, then "polluters pay" instruments, which suppose a constraint on the producers, are best suited to address the issue. The difficulty is thus symmetrical to that of "victim payer" or "beneficiary-payer" instruments. It lies in actually imposing and enforcing constraints on developers, as we have seen above.

Within this scope, one will meet similar problems to those already discussed in the case of land-buying or easement programs, offsets and banking being based on piecemeal deals. If ecological coherence and social viability are to be ensured, they are better operationalised as parts of a coherent planning regime. The potential to combine banking schemes with more holistic, systematic land use planning, setting aside corridors for rehabilitation by biodiversity banks appears intuitively appealing. It points again to high levels of administrative presence and load as a condition for the meaningful use of offsets and banking.

A last aspect of the scope that deserves discussion derives from the necessity of a firm cap for offset and banking to operate properly. As we showed, this requires legal and administrative conditions, but these rely in turn on high social demand and pressure for biodiversity. It is clear from our review that the demand and pressure play in two ways. On the one hand, they are the driving force for public authorities setting firm cap rules and effectively implementing them. But on the other hand, they can also play directly on development operators. Some of the firms involved in the activities that fall into the scope of offsets and banking are highly sensitive to reputational issues. For them, being able to report on projects with a neutral or positive biodiversity footprint can be a valuable asset (Bishop *et al.*, 2008). This is the motivation that underlies, for instance, the Ambatovey pilot project described above. Overall, whether it plays out directly through pressures on firms' reputations, or through public authority channels, the public pressure for conservation is also a scope-defining necessary condition for the development of environmentally effective offsets and biodiversity banking schemes.

4.4.3. *Organisational effects of trading for biodiversity*

Like all other tools, those based on trading biodiversity reinforce some specific actors.

Discussion with NSW scheme administrators highlights the key importance of a brokering sector, that is parties with an economic interest in the sourcing of biodiversity credits and the creation of functioning markets across ecosystem types. Offsets and banking effectively require the involvement of a whole range of organisations and specialised professionals. Finance, law and the overall brokering are all required by the complex negotiations of offset and banking schemes. If these develop to any significant scale, this will trigger the emergence of new operators and specialties in the biodiversity fields. Communication and participation professionals are also likely to have to get involved, both because of the reputational and acceptability issues in the use of such tools. Ecological assessment and ecological expertise will also have to be raised to levels higher than are currently available, because compensation-based schemes are much more complex and ambitious than impact assessment. A recent review of the Victorian biodiversity banking scheme found that there is currently a paucity of experienced support services from consultants or other private sector parties, which reduces the confidence expressed by market participants in achieving value through trade. In a similar way, the particular skills of ecological engineering are not widely held and are at the core of the viability of both offset design and then practical function and implementation of biodiversity credit creation.

An extended role of the regulating authority, which is part of the government, is also essential for the deployment of these tools. Its two roles are indispensable: "*to enforce compensation obligations on those creating debits, and to set the rules for establishing equivalence between those debits and credits*", (EFTEC and IEEP, 2010 p. 141), and the success of the system depends highly on the quality of playing these roles. See the example of the red-cockaded woodpecker, provided by ten Kate *et al.* (2004). Before the banking system, many incentives induced landowners and developers to take decisions adverse to the protected woodpeckers by not informing of their presence, or reducing their habitats until they disappeared, etc. The US Fish & Wildlife Service designed management plans through which landowners and developers collaborated, and were rewarded by (1) having their liability reduced if the woodpecker disappeared, and (2) being able to sell credits if the number of woodpeckers increased compared to the baseline. *International Paper* took advantage of the system to successfully create woodpecker habitats and then be authorised to cut forests where they were also present but in less favourable habitats.

Because a strong and powerful administrative system has to be deployed to rule an offsetting system, such an administration would have means to effectively implement all the other instruments as well. Again, offset and banking are no substitute for any other tool in the biodiversity toolbox. They are tools that require that all the remainder of the toolbox is correctly used, and then they can be used to provide some additional benefits for biodiversity and for the economy.

The regulating role is not provided only by setting up rules and laws. It is also by defining, through expertise and science mastered by dedicated agencies, the environmental references against which compensation will have to be designed and the priority areas in which compensation will be needed. However, conversely to what is often written in the literature devoted to offsetting and banking, it is not true that creating appropriate regulations is a sufficient condition to establish an exchange system such as a market for quotas, credits, etc. For instance, Entreprise pour l'Environnement (2009) provides guidance on compensation for business, and describes the case of compensation only by mentioning the law and directives contexts. In sum, it means that action is assumed because the environmental laws require them. The developments above suggest otherwise. First, the very adoption of the rule is certainly not a merely administrative and technical process. It involves conversely many political criteria and decisions. Second, acceptance by populations, judgment by justice, enforcement by administration, reputation of firms, definition of equivalencies, definition and re-assessment of caps… all require social as well as political interventions, all along the process of biodiversity offsetting and banking.

Hence, offsetting is not only based on a strong relation between law and market, but also on a strong link between the politics of rule-making and the politics of implementation.

Conclusion

This family of ETBs derives from the idea that managing biodiversity units, such as areas of habitats or populations of animals, could be eased by introducing the possibility to trade those units rather than enforcing uniform and rigid constraints, which would require a heavy and inefficient administrative load.

The history of these approaches tells a lot about their nature. Offsets are related to environmental impact assessment. They are the "last phase" of an authorisation process, which deals with residual impacts and allows a loss in exchange for a gain. Banking of biodiversity credits is rooted in the practice of tradable quotas, which are based on an administration having imposed some kind of constraint on access to biodiversity units, and allowing the trade of those rights to avoid uniform restriction, ease the system and make it less costly for firms.

As we see, originally, *both mean flexibility added to a system of constraints.* Limiting access to biodiversity creates some kind of scarcity, which then is managed by a market.

This is consistent with a classic version of the relationship between administration and market, the "de-centralisation" or "delegation" version: the role of the State and all institutions is to create an appropriate framework, in deciding overall political objectives (here, the "cap" to be imposed), designing rights (here, the definition of biodiversity units and conditions of access), and enforcing those rights (guarantee of rights and contracts by justice and legal force). Once this framework is established, the market is then supposed to do the rest and, if its forces are set free, distribute efforts and costs in the most efficient way, with reduced transaction costs and thus administrative burden.

Our analysis of these ETBs in use has brought some distance with a position that would adopt or reject this position wholesale, based on principles. It has however shown a reality somewhat different from the model of de-centralisation or delegation. Contrary to an organisation by which a framework is set up first by politics and legislation, policy, regulation and administration, and then left to the mechanical functioning of the market, we have seen something like a "lasagna" of administration and market. A first layer of institutions seems actually indispensable to express requirements and objectives, at least partly and provisionally, notably in terms of what is acceptable and unacceptable in terms of biodiversity loss, in defining units, access rights, etc. This setting up is nonetheless made by politics, in which of course the market forces and interests have their word, since they are taken into consideration in all stages of policy-making. Where and when decision-making has produced a clear objective, and is ready to enforce it through constraints, the market can then add a layer of exchange, adjustment, etc., which introduces flexibility. But, to do so, another layer of administration is indispensable, as the analysis "in use" made clear: exchanges have to be assessed in terms of equivalencies; impacts have to be monitored; credits have to be granted; objectives have to be re-assessed; conflicts have to be ruled; rights re-affirmed; offenders prosecuted; etc. This again requires much political deliberation (for instance, for redefining the cap, or in putting more or less pressure on offenders through pressure on control administration...). And the market can keep going and play a role only if this second functional layer is effective. In sum, whereas these ETBs are sometimes seen as typical market-like instruments, with brokers, speculators, exchange rates, etc., the "good" at stake (biodiversity) is one that is manufactured through administrative effort and needs constant maintenance by the administration. Exchanging such a good is, even more than for natural goods, a profoundly intertwined relationship between market and administration, and this interlacing has to be sustained indefinitely. This is of course a classical finding of institutionalists, but it may prove stronger in that case, and it leads to contesting an approach that would oppose a "command-and-control" approach with a "market-based" approach: command and control are needed at all stages of offset and biodiversity banking, which could be described as a "more command, more control and some market" approach to halting the loss of certain types of biodiversity assets.

If, then, "biodiversity trading" ETBs need a strong and competent administration, strong and enforced policy objectives, and a continuous intertwining of them with market forces, what would make them a solution to achieve a better biodiversity policy, rather than a new definition of the problem to be solved? If laws are passed and transformed into regulations that have the power to impose real constraints on other competing policy objectives, if the administration is adequately staffed to monitor, assess and guarantee the process, if decisions are enforced by justice, in those conditions, which instruments would be unsuccessful? What benefit would stem from adopting an ETB rather than other types of instruments such as regulation, protected areas, species conservation programmes, etc.? Two partial answers have come to light: (1) "trading" ETBs are better suited than other instruments for situations where threats to biodiversity are driven by high profitability and thus high opportunity costs (and on comparatively small areas); (2) by offering a means to enforce strong decisions with flexibility and consideration for the market interests, a trading system could make the decisions easier and favour the policy process. However this has not occurred very often yet.

Overall, offsets and biodiversity banking take on their full meaning as tools for biodiversity strategies that rely in an essential way not only on conserving some areas, but also on deploying new or restored areas for biodiversity. To be generalised, this will require the emergence of re-enforced administration in the planning and environmental areas, and of a whole sector of professionals able to engineer the redeployment of biodiversity in all its dimensions, from the ecological and technical to the financial and legal.

Those ETBs offer a potential for use, which can turn to be useful resources as well as enduring trouble. Like any medicine, those possibilities should not be considered indiscriminately. They will require the careful exploration of their conditions of use: from the "chain of authorisation" to the firmness of the cap, going through the links between market and administration, the types of threats, the technical equivalencies and the means available in terms of staff, political support and justice decisions.

5. Paying for Economic tools seen from the user's perspective: five organising questions

Can economic tools transform our ability to conserve and restore biodiversity? Can tools transform the trade? In the first four chapters of the book, we examined different compartments of the ETB toolbox and how the tools in them contribute – or could contribute – to better biodiversity management. In other words, we adopted an approach that started from the tools to examine the trade. But having a good grasp on the possibilities offered by the tools is not all it takes. The relation between tool and trade goes both ways. Two French proverbs take opposite perspectives on it: "Good tools made good tradesmen [18]" claims the first; "To the bad tradesman, there is no good tool [19]" responds the other.

We will now, and for the rest of the book, make a turnaround from the familiar tool-centred approach to a trade-centred perspective. Instead of looking at ESV, or PES, or buying land-rights, or biodiversity banking, and asking how management and policy contexts affect their use and effect, we shall henceforth start from the wider perspective: "How do we organise ourselves to conserve and restore biodiversity?" and reexamine from there how economic tools can or cannot help.

In this chapter we will take the time to purposefully turn the table around and effect a transition from the tool-centred perspective to the perspectives that start from overall strategic issues about biodiversity. (1) We will start by taking stock of the issues about the use of ETBs that we have identified in our review of literature and practice in the first four chapters. (2) We will then show that such questions all revolve around the fundamental problem of organised action in favour of biodiversity. The essential challenge is to understand how we can organise ourselves better to manage

[18] "Les bons outils font les bons artisans."

[19] "À méchant ouvrier point de bons outils."

biodiversity and in this context, to see whether and how ETBs can be used in relevant ways. (3) We will start on this track by examining the framework used by the TEEB report to represent the wider picture of our decisions and actions for biodiversity. (4) We will then show that to move forward, we have to invest on two different levels. We must intensify initiatives for incremental change, *i.e.* be more precise in how we use the tools in the situations and policy frameworks as they are. And we must seek deeper change, *i.e.* identify the limitations of how action for biodiversity is currently framed and look for ways to move beyond these limitations. Deeper analysis of utilisation issues can help us move forward on both levels. (5) The chapter will conclude with a set of five organising questions, each one of which will be addressed by one of the five last chapters of the book.

5.1. From tools to trade: pending questions

As we reviewed the literature, interviewed practitioners and reflected on our own experience, we read and heard over and over again that the use of ETBs and its outcome depended on context. The next challenge, then, is to find ways to analyse contexts of use, taking into account their huge variety and the great number of issues that are relevant to the use of ETBs. We need to be able to point to key features and to how they affect the use of ETBs. What do such analyses imply? As a first step to answering this question let us start by revisiting the review of ETBs in the first four chapters and by laying down the main questions they raise about contexts of use.

5.1.1. *Ecosystem services valuation: the need to really understand decision-making processes*

Concerning ecosystem services valuation (ESV), the need that arises is to understand decision-making processes about biodiversity. We have seen in chapter 1 that there are several extremely different concepts of what should be expected from ESVs. But whether one sees them (a) as providing a key criterion for decision-making, (b) as an external critical view on the lack of rationality of current policies, or (c) as a valuation process that contributes through advocacy to the more general decision-making process, it is important to understand how the decision-making process itself actually works and what can reasonably be expected in terms of getting better biodiversity outcomes out of it. Understanding decision-making processes implies several sorts of questions like the following:

Who is involved in the decision-making process, and in what role?

Is there one (or are there several) clearly identifiable decision-maker(s)?

On what basis are decisions made – *e.g.* do they approximate rational choice, do they rest on political confrontation, are they dominated by rigid bureaucratic procedures, or are they highly contingent?

How is information circulated – in what form, by whom and along what circuits?

How is information used and transformed by participants in the decision-making process?

In complex decision-making processes, influence is distributed amongst numerous actors; how does influence operate and what part can ESV play in such influence?

These are just some of the questions that are part of analysing decision-making processes, but it is clear from the start that there is not one single, nor any simple way of approaching them. Very different readings of the decision-making context are possible. The issue is to be clear about what one needs to understand, and how one will go about it – *i.e.* how one will select and organise the few questions that are most relevant in a given situation, or to follow a given systematic approach. But before we turn to that exercise later on in the book, let us continue our review of questions.

5.1.2. *Payments for ecosystem services*

If we turn to payments for ecosystem services (PES), they are transfers of resources between social actors in exchange for practices that contribute to ecosystem services. Most case examples, far from being simple, decontextualised payments, show such transfers to be a part (maybe sometimes a significant part, but still only a part) of a system of relations and deals between these actors, involving a variety of transfers of resources and of reciprocal influences, obligations and commitments. The central utilisation-related question is now to analyse this web of intertwined interdependencies and strategies, to which one plans to add one more transfer, one more deal – the PES. As we reviewed PES cases and literature in chapter 2, we underlined the importance of asking questions such as the following.

Who is the "buyer"?

What does the payment exactly buy?

Who is the "seller"?

And how and through whom is the transaction negotiated?

We saw that answers to such questions varied widely from one type of PES to another and from one case to another on the ground, and that differences in any of these aspects of PES schemes can have major consequences on their biodiversity outcomes. These four questions about the more or less direct aspects of PES deals, however, must be completed by others, addressing the wider organisational context within which the PES deal takes place.

What are the existing relations between the "buyer" and "seller"?

What kind of already existing "deals", be them cultural, social, legal, political, economic (including pre-existing subsidies and payments), etc., could a PES scheme either complement or disturb?

What power leverage does the one hold over the other? Are there third parties to the deal who are in a position to play a decisive role in helping or making the deal? In guaranteeing or obstructing its implementation?

What is the role of the monetary dimension in the relations between the actors in the specific context of a given PES deal?

These questions indicate that such understanding of context as is required, for biodiversity-relevant PES deals bears both on the particulars of the deal itself ("buyer", "seller", transaction and service bought) and on an organised analysis of the overall situation within which the business of managing the ecosystem and its ecosystem services is conducted. As any PES deal is only one part of the overall management of an ecosystem and its associated services, we now need explicit conceptual tools to understand these overall management situations.

5.1.3. *Buying land or land-based rights*

On turning to ETBs based on buying land or land-based rights, another set of questions about contexts of use has emerged from our review. Even in countries where land property rights are entirely or mostly dealt with through the market, we have seen that deals on rights for conservation and ecosystem services could not be made, or understood, independently of legal, fiscal, social and political contexts and processes, which condition both the goodwill necessary for such transactions to take place and the market conditions themselves (for instance as influenced by tax policies). The dependence on such non-market dimensions of context is a fortiori crucial too in those countries where land ownership is only partly allocated through the market, *e.g.* through regimes for the allocation of rights on public lands, or formerly public lands, or through regimes of community rights on land. Biodiversity and ecosystems depend crucially on land use. Land use depends in turn (1) on property rights (the fulcrum on which ETBs based on buying rights base their leverage), but also (2) on the technical-economic conditions that make alternative land-uses and practices feasible or not, attractive or not, and (3) on the wider context of history, of social systems, of power relations and politics, of public policy, subsidies and macro-economics, and of other aspects of the overall game that determines the uses of land over entire areas at all scales, from a given rural landscape to entire regions and countries, and across countries. Questions like the following then arise about the contexts of ETBs based on the purchase of land or land-based rights:

Who is the operator of the tool: a private entity, a public body?

To what extent is the allocation of land the result of private deals, of political regulation, or of intertwined processes involving both?

How are heterogeneous dimensions of land property and land use, such as land-based technical-economic production systems, history and social needs, efficiency and justice, etc., articulated in analysing the context of land-rights-based conservation deals?

In short, such deals have to be understood in the specific context of each land tract that is bought, but also in the contexts of how the use of land is negotiated – or struggled over –, at wider scales, between social actors.

5.1.4. *Offsets and biodiversity banking*

Turning finally to offsets and biodiversity banking, the first central finding that emerged from the literature and cases was that such schemes can work only on the basis of limits to the loss of certain ecosystems or biodiversity components. These limits have to be set so firmly, and to be implemented in such an effective way, that carrying out the offsets or trading credits becomes the best option for operators on the ground. In order to hold, such limits have to be backed by public pressure on the licence to operate of companies in a given industry. In most cases, this has to be consolidated by law and by effective law enforcement. This can only be the result of an overall state of affairs where social actors have shown an effective capacity to establish and implement a collective commitment to halting biodiversity loss and making the required changes in technical and economic practice. In other words, biodiversity banking and other such sophisticated tools can effectively operate only in contexts where wider problems of setting limits collectively to biodiversity loss and ecosystem degradation have been resolved or are being very actively resolved.

This is the kind of ETB that demonstrates most clearly that the tool cannot replace the trade; it can make the trade easier and more efficient, but it provides no substitute for the overall capacity to take charge of given biodiversity issues. If biodiversity banking is based on limits set to specific aspects of biodiversity loss, there is no dodging questions like the following:

How are we able to set limits on projects that some of us want very badly?

Can we negotiate such limits all together, or can they be established and enforced through the mobilisation of some actors with special powers or motivations?

What is the part of negotiation and what is the part of coercion in our dealings with such problems?

What is the basis on which the limits rely: ethical principle? rational calculus? a power struggle?

5.2. From trade to tools: a question of organised action

As we shift through our review of the questions that are raised from the ground, and from the literature, about the use of ETBs, we look for a common theme that could help organise these questions in ways that may at the same time be clarifying for the practitioner and support the mobilisation of social science, humanities, politics and management scholars in support of biodiversity management. The emerging questions revolve around three questions. Who operates the tools? Who deals with whom and how, based on the tools? How do these deals contribute to our collective ability to act effectively on biodiversity issues? To us, these questions pertain to a problem of organised action. ETB use is organised action in the sense that it is not (or only exceptionally) action by one agent, but action involving several agents, connected between themselves in more or less structured ways and interacting with others within partly organised frameworks. Its outcome depends in large part on the structures and processes through which what each one does (in intended or unintended ways) links up with what the others do – that is, on organisation. As we turn the perspective around and examine the tools from the perspective of the trade, we can sum up the trade we are interested in: organised action for biodiversity.

The phrase may sound very general and abstract, and it may seem to some readers that, by moving from specific field situations to such a wide problem, we might have gone backwards, rather than forwards, in terms of actual use of ETBs. We have already made the case, through our review of the difficulties met in ETB use, for a reflexive step backwards, one that will help us return to the use of ETBs with a firmer strategic purpose and a renewed focus on making them work. Why do we think organised action to be a particularly relevant framework for the exercise?

5.2.1. *A cross-disciplinary framing of the issue of ETB use*

First, it is wide enough. It subsumes issues and patterns that run through alternate disciplinary perspectives on biodiversity field situations. Whether we examine these in terms of policies, of management systems, of politics, of administration, etc., the common core of collective action issues is always there. It points to some of the most important issues that are present across disciplines, without forcing us to choose between them. It also runs across various theoretical perspectives, which we shall see is very useful, since the analysis of complex managerial and political systems and processes is an arena of competing perspectives, very different from one another, each one of which (at least, each of the more interesting ones) being able to make a specific contribution. It is particularly enlightening, and practically useful, to see how different theoretical approaches differently contribute to our understanding of collective action for biodiversity.

5.2.2. *A clearly focused framework with a deep academic background*

Second, the problem of collective action for biodiversity is precise enough for our purpose for a deeper look into the foundations of using ETB tools to act for biodiversity. It supports a set of essential, related questions and concepts that can guide us quite far into the investigation, both theoretical and empirical, of situations where we want to resolve a biodiversity issue. Who is supposed to act? What does action consist in? What kind of relations and deals does action for change entail? These are questions that reach very far both in relation to field cases and in terms of the theoretical resources and developments they invite. One condition, however, is required for us to mobilise their potential for clarification: that we resist the temptation for quick, stereotyped answers. In our view, much of the current difficulty in dealing seriously with the use of ETBs comes from thinking either: "we know how decisions are made (meaning: by a decision-maker who needs better information, by a community whose members have only imperfect understanding of the values of ecosystem services, by markets that would need specific signals to correct particular market failures, under the sway of public opinion and media, etc.), let's move on to providing better economic tools for this decision-making;" or "come on, people have been struggling with this issue for years without success. I know what the block is (meaning: ineffectiveness of regulation, lack of private property rights, wrong signals sent to producers,…); let me try tool x and you will see!" We seriously need to suspend premature diagnostic. How environmental management functions and environmental decisions are made, are sophisticated and controversial questions in their own right. They deserve in-depth treatment. And they receive it to a large extent through a vast (and rather heterogeneous) body of literature, and through intense debates, involving both academics and practitioners, that have been going on for at least four decades. Practitioners and academics involved in ETB use, however, as they have focused mainly on ecology, economic theory and on the pragmatics of environmental policy, have engaged such arenas of production and discussion of ideas only to a limited extent. In view of the difficulties encountered in the use of ETBs, a larger set of conceptual frameworks, of theoretical models, of alternative perspectives should now be mobilised in a very explicit and detailed way to take a wider and deeper look at our difficulties in addressing biodiversity issues, in particular through the use of ETBs. The problem of collective action, through the set of questions – at the same time generic and precise – that it sustains, seems to us particularly appropriate in guiding this effort in a precise manner, without forcing us to adhere prematurely to one theory and its diagnostics of ETB use issues.

5.2.3. *A framing of use issues with high practical value*

Third, the problem of collective action has clear practical value. Its set of questions, such as "Who is supposed to act? What does relevant action consist in? What kind of relations and deals does action for change entail?" – are ones each ETB user has to carefully consider and answer in practice. They provide both strong leverage for a critique of routine practice, and they are directly oriented towards action. Their critical leverage comes from asking questions to which we thought we already had the answers – when maybe our answers were not particularly good, considering the results. So they help us to reconsider routine, premature diagnostic. And at the same time, they guide us in looking for actionable answers. This reconsideration of situations leading to actionable answers is the essence of strategic thinking in practice. To support it, it is particularly useful to have (a) a clear, organised, set of guiding questions and (b) an enlarged repertoire of diversified, alternative answers to such questions. This is precisely what we shall try to provide through the rest of the book, so as to help expand the current strategic thinking on ETB use.

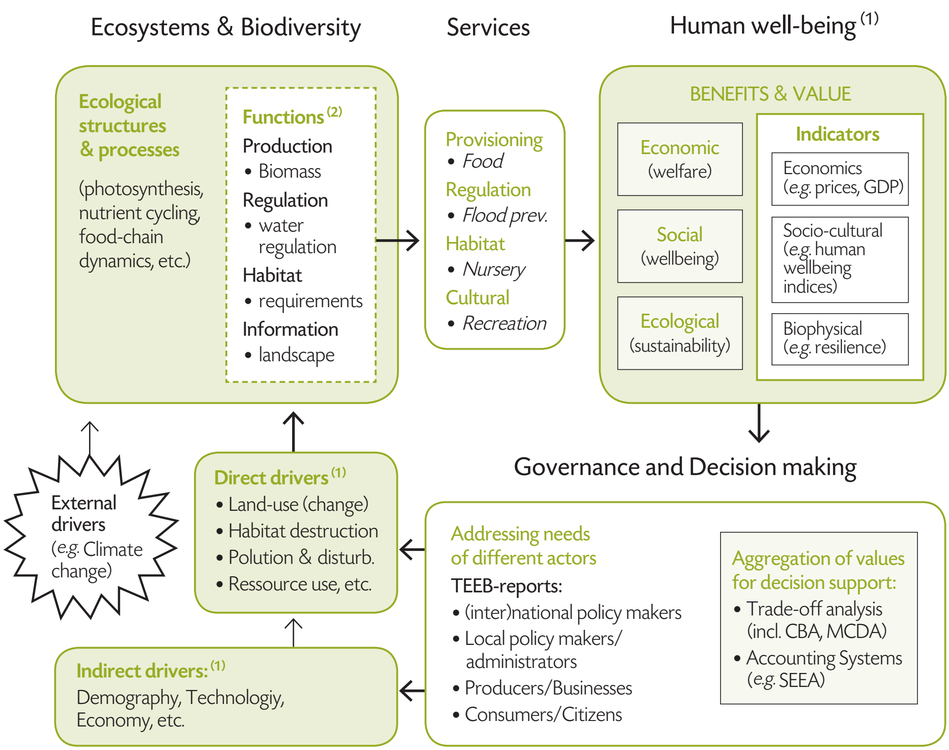
5.3. Engaging the most widely-shared concepts and framings about the use of economic tools for biodiversity

The logical place to start is through an examination of the standard set of questions and answers that represents current strategic thinking on ETB utilisation. Fortunately, a systematic exposition of such widely shared strategic thinking is readily available. The TEEB report provides just such an exposition, or comes as close to it as can possibly be, so we shall use it here for our discussion on state-of-the-art reflection on the stakes and contexts of ETB use.

5.3.1. *The TEEB framework as a reflection of widely-shared thinking on the contexts of ETB use*

We shall focus on the carefully constructed framework in which the authors of the first chapter (de Groot *et al.*, 2010) of the main volume distil the best from various state-of-the-art frameworks in the field and organise an overall picture of the contexts in which economic tools can be used to manage ecological issues (Sukdhev *et al.*, 2010) (cf. Figure 4).

Figure 4 *The TEEB conceptual framework*



(1) The four shaded boxes coincide with the overall MA-Framework.
(2) Subset of ecosystem processes & components that is directly involved in providing the service.
Source: Kumar, 2010.

The framework's ambition is to "link ecosystems and human well-being". It presents a view of the fundamental organisation of action to manage ecosystem services and biodiversity issues. In a nutshell, (a) better information on how ecosystems provide ecosystem services, on the benefits and values these services provide to society (upper half of the framework), and (b) information on how the services, benefits and values are affected by negative impacts from drivers (left half of the framework), is to be used (c) by actors involved in governance and especially decision-makers to take action for change in those drivers that cause loss of ecosystem services (lower half of the framework).

This model of the action system and contexts in which ETBs are to be used quite relevantly reflects state-of-the-art reasoning that underpins most of the ETB field. First, the TEEB report is clearly committed to dealing in depth with the issue of use and the quest of the authors for concrete use in real-world contexts is apparent throughout. This comes out most obviously in the large volumes that the report dedicates to specific groups of users – national and international policy-makers, local and regional decision-makers, business and enterprise. And the framework is not just an academic aside to that pragmatic project, but a carefully constructed synthesis of both practical and academic treatment of ETB use. Second, the report has mobilised a large number of well-respected authors and reviewers from around the world. It has involved a thorough process of feedback and debate and the text clearly reflects the ideas – sometimes divergent, sometimes convergent – that dominate the field's state of thought on issues of ETB use. Third, the report is quite recent and appears to be a relevant starting point for our exploration of new avenues for further development of ETB use and of research on ETB use. Finally, our interviews with experts and practitioners of ETBs widely converge with the points of view expressed in the TEEB report. Diverse as they are, they all remain within the range of views articulated by the report. And even those that are somewhat critical of the report point to its limits, but do not put on the table any clearly formulated alternative treatment of the issues studied by the report.

The most frontal critique of TEEB we have heard in our interviews was expressed by a consultant with a long experience in applying economics to environmental policy issues in the UK. "TEEB work is work that hasn't changed in the last twenty years. I'm concerned that very little intellectual effort has gone into these frameworks. The framework hasn't changed; the evidence has improved a little." The interviewee goes on to note that the Oxford Economic Review's January 2012 issue [20], devoted to the same issue, has not identified new approaches either. While we obviously converge in perceiving a strong need for breaking beyond the current state of the art, we would rephrase this interviewee's observation in a positive way. The TEEB report is a thorough and well-thought-through summing up of collective wisdom and research accumulated over the last thirty years, a synthesis of a state of the art beyond which it is both necessary and difficult to break. Let us take a closer look at the framework in this perspective.

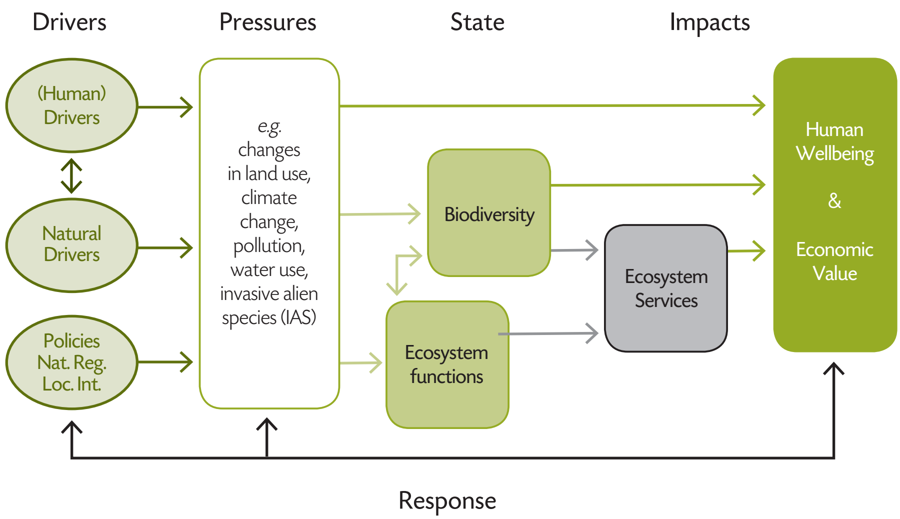
[20] See: http://oxrep.oxfordjournals.org/content/28/1.toc

5.3.2. The TEEB framework links together applied ecology, environmental policy and environmental economics

A most important aspect of the framework, in our view, is that it links together collective thinking that has been accumulated in three quite distinct fields of practice and literature: ecology as applied to biodiversity conservation and environmental policy, environmental policy design, and environmental economics.

1. As underlined by the TEEB authors themselves, the boxes connecting ecosystems, services, well-being and the pressures that affect them correspond to the framework of the Millenium Ecosystem Assessment (2005). This was in its own right a summing up of the collective thinking accumulated in the last four decades in several components of applied ecology: the study of ecosystems' structures and functioning (upper-left box); the whole part of environmental science that focuses on impacts of human activities (lower-left box); the effort to focus ever more on the benefits that people accrue from ecosystems (upper-right box). Even though the latter is sometimes presented as new, in fact it reflects a movement that has been guiding the conservation and environmental policy community for 30 years, since the 1980 IUCN World Conservation Strategy, which introduced both the call for sustainable development and the concept of life-supporting ecosystems, of which ecosystem services are the current offshoot.
2. The treatment of "governance and decision-making" (lower-right and left boxes) relies on a differentiation of sub-systems and on connections between them that follow the DPSIR (Drivers, Pressures, State, Impacts, Response) framework (cf. Figure 5).

Figure 5 *DPSIR framework*



Source: Adapted from Braat and ten Brinck, 2008.

Although there may be minor mismatches in the correspondence, on the whole the "Indirect drivers" of the TEEB framework are the Drivers (causes) in DPSIR, the "Direct drivers" are the Pressures, the "Ecosystem & biodiversity" are the State (condition of the environment), the "Services" and "Human well-being" are the "Impact" (which in the DPSIR covers both ecological and well-being impact, just as in the upper-right box of the TEEB framework), and the "Governance & decision-making" is the DPSIR Response. Evolved from the Pressure-State-Response reporting framework implemented by OECD in the early 1980s, the DPSIR framework is currently widely used to organise information (for instance, statistics) and diagnostic on environmental issues by major national and international institutions (see figure 5, following Braat and ten Brink, quoted by Patrick ten Brink *et al.* (ten Brink 2011, p.86).

This reliance on DPSIR-reasoning is especially important to note here, since it does constitute a theory of action on the treatment of environmental issues (of biodiversity issues in the TEEB context and in the context of our own work here). It has been adopted as the main theory of action in the TEEB report – and it has not been particularly challenged (explicitly or implicitly) in our interviews with experts and practitioners in the ETB field. This suggests that it is particularly relevant for consideration here as a springboard for our further investigation, through the rest of the book, of possible alternative theories of action for biodiversity

3. The third mainstay of the TEEB framework, of course, is the accumulated wisdom of the economics of the environment and biodiversity. We may note that the diversity of view in the report does reflect the diversity of views within that field, as reflected by the tension between the "environmental economics" and the "ecological economics" perspectives. In the framework, economics does not have large boxes of its own, but appears as a component of the human well-being, of the decision-making and of the "drivers" boxes. In TEEB's perspective, these do indeed sum up the various contexts of action in which the economics of biodiversity are to be used.

5.4. Are we considering tools for incremental or for fundamental change?

But before we examine in more detail how various roles and use situations of ETBs are seen in TEEB's perspective, we have to stop for a moment and note that the state-of-the-art approach of ETB use, and the TEEB framework as it reflects it, confront us with a dilemma: should we strive to get more done within these current framings of action for biodiversity, or should we rather break out of the mould as it is too narrow? In other words, should we press ahead to do our best under the constraints that weigh currently on our action in favour of biodiversity, or should we directly address such structural constraints because there would be no adequate solution to biodiversity issues within them? In the following two sections, we will examine successively each branch of the alternative.

5.4.1. *A call to arms: we can do much more within the current framings of environmental policy*

If we accept to examine the use of ETBs within the TEEB framework, it is immediately apparent that there are essentially two roles for ETBs. (1) Ecosystem services valuation allows assessing in economic terms the effects that changes in ecosystem services have on human well-being (upper part of upper-right box). Combined with other systematic indicators of social and environmental values of ecosystem services (right part of upper-right box), and with other economic data (right part of lower-right box), this provides information for the support of decision-makers. (2) These can then use "money on the table" economic tools as part of their interventions for change of indirect and direct drivers affecting ecosystem functions and services (black arrows pointing at two lower-left boxes).

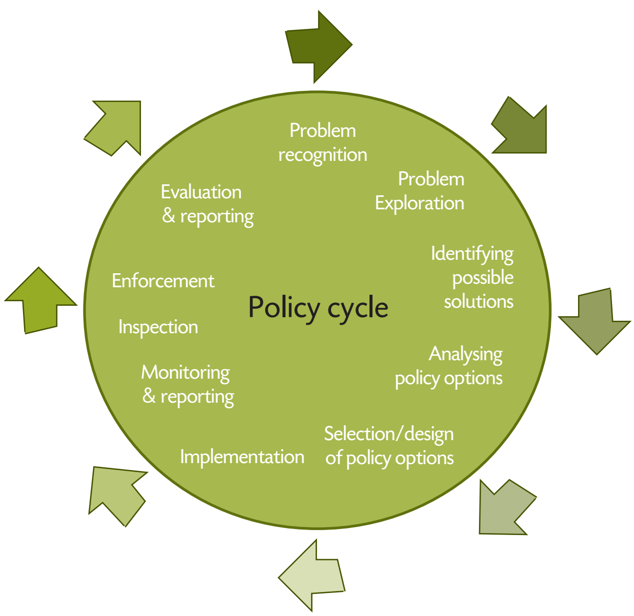
As a description and analysis of the roles of ETBs and of their context of use, the framework remains, however, extremely vague. On both types of uses, TEEB translates views that currently prevail in the field into a global-scale call to arms. It focuses on showing biodiversity stakeholders that they should act more decisively on biodiversity, and that they can do much better by using ETBs more to enhance state-of-the-art biodiversity management and policy. TEEB's fundamental logic is that stakeholders and policy-makers involved in biodiversity issues know best their practical situations and that if one adds to that – as is the whole point of TEEB – a keen awareness of ETBs and their potential in practice, then they are going to use them more and to better effect. To sum up, its way of advancing the detailed analysis of ETB roles and use-contexts is case-based, eclectic and user-driven.

The alternative – in fact, complementary – way to contribute to the detailed analysis of ETB use situations and strategies is the one we are proposing in this book. We do not think that awareness of the available economic tools is the main limiting factor blocking their effective use in favour of biodiversity. In our view, current limitations rely more on a combination of (a) diagnostics of biodiversity management situations that are often based on superficial or misleading notions and conceptual models and (b) notions of how ETBs effectively work that are based on abstract economics that do not reflect their actual operation and potential as management and policy tools. What is needed, then, are much clearer conceptual models of biodiversity management and policy challenges and of how ETBs actually work. Such models will allow practitioners, experts and researchers to break out beyond current limitations in mobilising ETBs to complement the existing array of biodiversity management and policy tools. The numerous questions arising from the first four chapters, and summed up at the beginning of this chapter, point to the intensity and scope of what should be expected from improved diagnostics that would help carry the use of ETBs much further even within the current constraints under which biodiversity management and policy operate. Before we organise these questions and expectations in a more systematic way however, we must examine the limits of the state-of-the-art framing of biodiversity management and policies and see if we must, if we can, break out beyond them.

5.4.2. Time to reconsider how we can move beyond the current bounds of environmental and biodiversity management and policies

As it reflects the conceptual model of action that currently dominates the field of biodiversity management and policy, the TEEB conceptual framework also provides a relevant basis for a reflection on the limits of that model of action. A good place to start is by taking a closer look at the TEEB and DPSIR framework. Their main logic is one of corrective action and remedial intervention. This is well illustrated by the model of the policy cycle (see Figure 6) used by ten Brink (TEEB PM p. 85) in connection with the DPSIR graph to illustrate the use of information (for instance, of ESV) in acting on biodiversity issues.

Figure 6 *The policy cycle concept as used in TEEB*



Source: Patrick ten Brink.

This view of policy as proceeding in stages from problem identification to choice of action, implementation and evaluation of outcomes can be criticised as "linear, mechanical". It may be considered simplistic, ignoring the systemic, value-laden, political nature of issues and policies. However, it is deeply rooted in sophisticated, highly influential views of policy.

In terms of policy design and evaluation, it reflects the concept of public action as mostly organised through programs and projects, a concept which is very influential in North America and has rapidly spread across international policy arenas. It sees public action as best organised through separate programs, each addressing a given problem at a given time. In principle, this has the advantages of avoiding non-necessary action (action that does not address a clear problem), of organising implementation in a rational way (program design is guided by problem-solving) and of allowing for sound evaluation (program can be assessed against the problem to be solved). This perspective is very influential in the field of ETBs. The literature on PES, for instance, very often spontaneously envisages "PES projects" – *i.e.* the payment for ecosystem services is seen and managed as a distinct project, to solve a specific problem.

From the economist's point of view, such remedial action can be seen not so much as linear as marginal, in the economic theory sense of the word. The economic system functions as it does, and one is founded to reason action in marginal terms, *i.e.* by asking what additional intervention could improve the outcomes (for instance by improving ecosystem services that have economic value) in the most efficient way.

Looked at from the perspective of the environmental policy, this model sees environmental policy as remedial action, aimed at correcting or preventing specific environmental impacts of development, including impacts from other policies. Agri-environmental subsidies, for instance, aim at mitigating specific negative impacts of agriculture, for example by preventing conversion of pasture to maize. A major characteristic of this model of environmental policy as a series of remedial measures is that it does not challenge the deeper structure of production systems or sector-based policies. In that limited frame – which currently dominates the doctrine and practice of environmental policy – the remedial "pressure-response" logic of DPSIR provides an adequate concept of the main foundation of action. And so does the use of ETBs as seen in the TEEB framework, *i.e.* as tools for measures that can prevent or remedy losses in ecosystem services that would result from the drivers of environmental change.

So overall, the currently widely-shared view of action in favour of ecosystem services sees such action as additional, remedial, coming on top of, or beside, the main social, economic and development policy systems. This view sits at the convergence of the economist's marginal reasoning, of environmental policy's place as complementing and trying to inflect sector-based policies, and of the program/project-based concept of public action. So the roots of the concept of action reflected by the TEEB framework are deep. In practice, they reflect the very foundations of action for biodiversity as it is conducted today. In theory, they are strongly defended by the mainstream literature in policy evaluation, in economics and in the doctrines of environmental policies.

It is less surprising, then, that it appears difficult to move beyond them, either practically or intellectually. The necessity to do so becomes apparent, however, as soon as one becomes aware of the deep contradictions involved in this standard model of environmental policy and thus of intervention for biodiversity. If we take a closer, somewhat critical look at the TEEB framework, we can see an implicit connection: some of the most essential "drivers" that threaten ecosystem services are caused or boosted by public policies. Urban development, land use, technology, are not natural forces that could be envisaged independently of planning, of agricultural policy, of research & development policy, of development aid, etc. If we look at drivers that impact biodiversity and ecosystem services most on a global scale, the extension and intensification of agricultural production, the exploitation and clearing of forests, and unsustainable fishing, all supported by numerous and powerful national and international policies (regulations, funding, research support, etc.). Certainly there also exist some national and international (environmental) policies that attempt to mitigate these drivers and limit their impacts. But it would be hard to argue that the efforts that are invested in the latter (biodiversity conserving) types of policies are sufficiently greater than those directed at the former (biodiversity damaging) for us to be on track to halt biodiversity loss on a global scale. We are indeed caught in a major contradiction of public policies, one that the remedial model of action in favour of biodiversity cannot address adequately.

There is a logical paradox at the heart of environmental policy, and at the heart of serious reflection on the use of ETBs. Policies are requested not only to influence social, economic or technical systems to make them more sustainable, but also to change other, deliberate and powerful policies that are not sustainable. This leads us right to the core of the problem of organised action for biodiversity. If we must act to change our own actions, who is acting on whom? And what does that action consist in? This raises deep issues and we will need more than a linear model of policy as recognising and addressing problems on a case by case basis (as in the "policy cycle" and DPSIR models) to analyse them.

Through this critique of the limits of environmental policies, and of the conceptual models that most often underlie how they are analysed in the ETB field, we do not imply that the promoters and operators of ETBs would not be aware of the limitations of "remedial" approaches that would leave the wider framework of policies and the economic system unchanged. On the contrary, we are struck by the fact that a great number of operators and analysts point to the need for deep changes at the very core of the economic system and our policies. The TEEB report, for instance, proposes the following quote (Hansjürgens, 2011) "*Success will require two major shifts in how we think – as policy makers, as campaigners, as consumers, as producers, as a society. The first is to think not in political or economic cycles; not just in terms of years or even decade long programmes and initiatives. But to think in terms of epochs and eras* [...] *the second is to think anew about how we judge success as a society. For 60 years we have measured our progress by economic gains and social justice. Now we know that the progress and even the survival of the only world we have depends on decisive action to protect that world.*" This is not a quote from some activist, or academic critical thinker, but from Gordon Brown, at the time (2009) Prime Minister of Great Britain. Yet this quote plainly spells out the fundamental limits of program-based remedial approaches and calls for fundamental changes in how we operate our society.

However, once one realises the necessity of major changes in policies and in the economic system, one is led back to the fundamental issues of collective action. Who is going to act for such changes? What kind of actions can be effective? What kind of interactions with other actors can initiatives for deep change generate? And since success ultimately depends on such interactions, we crucially need a clearer, more firmly based understanding of how they may play out.

5.4.3. *Action for incremental or for fundamental change is still collective action*

So, considering fundamental changes raises similar issues about how we can act as does the effort to use management tools for more incremental change. In our view, the current challenges of biodiversity management, and of the use of ETBs as part of acting on those challenges, require that one analyse jointly the difficulties of projects for incremental change and the difficulties of fundamental transformations within a wider reflection on collective action. Let us successively lay down theoretical and practical reasons to do so, and then show that the field is indeed already deeply engaged in perspectives that – although with much ambiguity and sometimes awkwardness – join the two perspectives together.

Starting with theory, using the idea of "organised action for change" as our Ariadne thread allows us to benefit from the dual nature of "organisation" as a concept, embracing on the one hand the way we are organised (the way we are organised now because we have so organised ourselves over time) and on the other hand the process of organising (the process through which we act now to modify the way we will be organised further down the line). This dual foundation is of the essence in the context of solving biodiversity, and more widely, sustainability issues, because we are caught between two imperatives. On the one hand, any concrete action that we can do now, we can only do based on the way we are organised now. On the other hand, solving the issues clearly requires that we become organised differently. The deep tension between how our present organisation conditions our options for acting concretely, and the dynamics of changing this same organisation to get the outcomes we want, is at the heart of organisation both as a field of practice and as a theoretical concept. It is at the core of the dilemmas we face as we try to solve a biodiversity crisis that largely stems from the ways we are organised, and that we have to solve starting from within this (biodiversity-wise) unfortunate organisation.

This abstract formulation of our predicament echoes the overall practical malaise that is likely to appear when we reflect on how we do overall at managing biodiversity. The high level of mobilisation of all sorts of actors, the large number of public and private initiatives and the ever-growing presence of biodiversity issues in the media are impressive. But are their scale, their power, their strategies and the way they are organised on a par with the drivers that will have to be curbed if the biodiversity crisis is to be resolved? Can doing what we are already doing only more intensively and more efficiently, thanks to ETBs and other tools, make enough of a difference to turn around the trends that lead us to an ever more unsustainable use of ecosystems? Then if we look at the level of more specific biodiversity management situations, why is it that cases of good biodiversity management seem to remain comparatively few and far between so that we tend to present them as inspiring examples that should be replicated and generalised? Is it just because the solutions had not occurred to other people in other places, that they were using the wrong approaches, that they needed pilot projects to show the way (Pirard *et al.*, 2009) ? Or is it because effective action for biodiversity is so difficult that it succeeds only in a limited range of situations and fails in many others? In practice, on the ground, the difficulties of incremental action and of deeper change are closely knit together, and whoever takes practical action for change – in our case, biodiversity motivated change – has to take them up together.

Such reasons, both theoretical and practical, lead us to the position that the challenges of incremental action in the here and now and the challenges of deeper change have to be firmly held together and analysed jointly as we consider the challenges of collective action for biodiversity. As we take this position, we refuse to make a dichotomic choice between on the one hand doing one's best with the available tools in the situation as it is, and on the other hand, rejecting the managerial use of tools wholesale on the grounds that they are no more than diversions from deep, significant change. Practical action now is worth intensifying, but it requires that we become more and more lucid on the forces at play in the situations we face, and on the conditions for success, biodiversity-wise. As for deep changes, they are important mobilising goals, but they have to translate into tangible strategies for action now. We feel that the tension between the joint needs for deep change and for action in the here and now is widely shared in the biodiversity field.

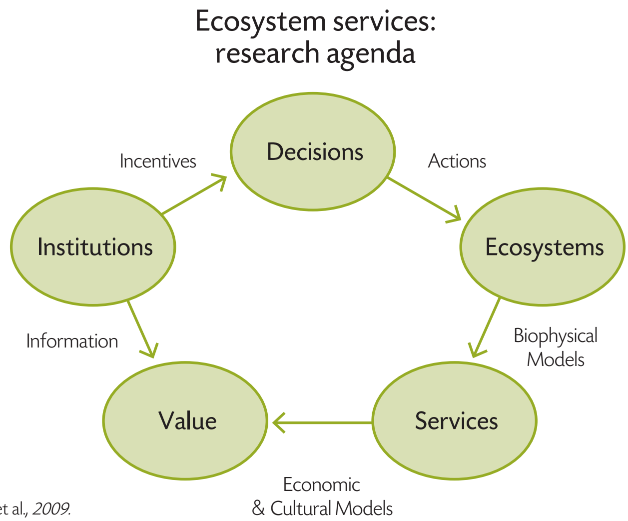
5.4.4. *Prospects sitting on a fence between the incremental and the radical*

A similar tension or ambivalence is clearly more generally apparent in the main proposals that are discussed in the biodiversity field for transforming policy and management: public participation and deliberative democracy, institutional change, and a shift in values and lifestyles. Let us examine them in turn.

Participation is often put forward as the most promising way to reconnect environmental action with society – as well as to mobilise society, from local to global on environmental action. Since the early 1990s, public participation has been at the centre of the agenda of environmental action and policy. It is now part of virtually each and every approach to environmental issues, in particular in the field of biodiversity. Strikingly, it is both put forward by mainstream actors of environmental policies and by radical critiques. The field of ETBs is no exception, and much of the literature on the use of ETBs and on how to help integrate them better in their social contexts does focus very much on participation. There are signs, though, that the support for participation as a central element of solutions to environmental problems has reached its peak and that its limitations as a tool for change become more apparent. Participation can be a channel for change; but it can also modify the procedures of planning or of development policy in appearance, without changing significantly the outcomes, leaving sustainability issues largely unresolved. At issue is the connection between participation and in-depth change, a connection which is rarely addressed clearly in the literature and in the practice of participation.

Institutional change is also a salient proposal in the biodiversity literature. Consider for instance, one of the figures used in TEEB to build up its conceptual model (cf. Figure 7).

Figure 7 *A model of the relationship between valuation and decisions*



Source: Daily et al., *2009.*

In this model, the connection between the tools ("information" and "incentives") is quite clear: they are used by institutions, but they can also lead to changing institutions. In the words of the report's authors: "information [on ecosystem services and their value] can lead to the reform of institutions and better decisions that ultimately improve the state of ecosystems and the services they provide to society" (ten Brink 2011, p.9-10)). The role of institutions in our management of biodiversity is clearly central. However, the idea of changing institutions, the ways by which "information [...] can lead to the reform of institutions", is clearly missing much in terms of a clear theory of action. Is the stake to topple existing institutions and replace them by new ones? If so, who can effect such change and how? Or is the stake to introduce additional institutions to the existing system, and if so, how is that to change the biodiversity problems stemming from the existing institutional system?

Finally the idea of the need for some sort of cultural revolution and change in lifestyle is also very present in the biodiversity literature. But is there really a clear and useful distinction to make between incremental and deep change here? Just as in the case of changing institutions, a shift in how we think or in our lifestyle requires concrete action; it cannot occur in a completely abstract, purely cognitive sphere. Appropriate forms of collective action are required if we are to change how we think, what we do and how we do it.

Conclusion: a set of organising questions to analyse and design collective action for biodiversity

The next five chapters of the book will be devoted to a systematic examination of concepts, models, theories that we can use for a deeper analysis of collective action for biodiversity, and of the various ways in which economic tools can contribute to such collective action. They will follow a set of five organising questions that articulate the multiple challenges and interrogations that are raised by the current use of ETBs and attempts to expand it. Let us quickly recapitulate the stages we have covered to this point and on the basis of which we shall articulate these questions. The first four chapters brought us again and again to the conclusion that better use of ETBs is highly sensitive to contingent contexts of use. Improvements in ETB use will have to rely on a better understanding of contexts of use and be part of more explicit strategies to solve biodiversity issues in those contingent contexts. This was a frustrating conclusion however, because it demands that one explain what exactly would constitute such better understanding of context, and such strategic thinking.

In this chapter, we started by showing that each of the first four chapters had delivered a series of more precise and pointed questions to guide the analysis of context and strategies involved in the use of various sorts of ETBs. We also examined – especially based on the TEEB report and on the interviews we made for the book – the state-of-the-art reflection that underpins the ETB field's efforts to improve the use and relevance of economic tools to solve biodiversity problems. This now leaves us with two complementary challenges: (1) to do more and better within the current frames of thinking and action, we need much more precise conceptual models of biodiversity management and strategy; (2) we also need to be able to move beyond – and to start with, to think beyond – the current underlying frames of biodiversity policy and management and for this, we need strongly built conceptual models that would provide alternatives to the currently prevailing framing of biodiversity issues. An essential conclusion though is that rather than being opposed in a black-and-white manner, as some might do taking opposite sides, both challenges are inseparable in the field and raise concurring questions. They both belong to a wider problem, a problem of collective – or of organised – action. As a result of these successive stages, we propose the following five organising questions through which we can approach the problems of organised action involved in using ETBs to treat a given biodiversity problem.

1. Can we be more explicit about *who exactly is trying to solve the biodiversity problem?* The fuzzy assumption that biodiversity problems are common problems shared by all, the fact that in the field there is a great diversity, but also a great ambiguity about who is acting for biodiversity, feeds a certain confusion on who exactly is in charge of solving each biodiversity problem and who is using ETBs in trying to do so. In chapter 6, we will propose a specific framework to analyse this question of agency, both for theory and for practical diagnostic.
2. Can we *solve biodiversity problems by agreeing on new rules and institutions?* The setting of new rules is a mainstay of environmental policy and management. As we have seen in chapters 1 to 4, effective implementation of ETBs often crucially depends on new rules, effectively implemented. In chapter 7, we will explore some of the theoretical and practical challenges of rule-making (and institution-building) by using and discussing common-pool resources theory.
3. *Can we clarify the contradictory values* that compete and clash in the treatment of biodiversity issues? Clashing values – for instance the insistence on economic efficiency versus the critique of commodification – may be one of the most immediately apparent problems in the ETB field as a whole. In the field, the clash takes more complex and varied forms; it can be more implicit, but still affect ETB use or effectiveness in deep and diverse ways. In chapter 8, we shall see that using and discussing Boltanski and Thévenot's justification theory can provide powerful and relevant resources to improve our understanding of these challenges and of how to deal with them.
4. *How can stepping outside the impasses of the current situation help us think about technical and political innovation?* The challenge here is that we live in a rapidly changing world. Many biodiversity issues seem to be in an impasse, considering the pressures and contradictions of the world as it currently is (be it locally or globally). To solve such issues, we have to consider transforming situations deeply; we also have to consider that biodiversity issues will have to be dealt with in situations that will have been transformed themselves (for instance through changes in technology or in political relations). In chapter 9 we will mobilise Michel Callon's "translation" theory and Bruno Latour's "politics of nature" framework to help us intellectually (and hopefully, practically) move beyond the ruts of technological and political relations as they are and consider innovative arrangements in relations between society and nature.

5. How can we *analyse resistance to biodiversity motivated change and understand the position and strategies* of biodiversity advocates as they confront the considerable powers involved in biodiversity issues? Solving most biodiversity problems involves significant changes in the way some spaces are managed, or some goods are produced. This requires that some operators engage in strategic action in favour of such changes. But this will most probably – as field experience on biodiversity repeatedly shows – elicit resistance to change. Strategy and power relations are a reality of using ETBs in the field, be it to change policies, or to act on local issues. To help analyse them in a structured and systematic way, we will use strategic environmental management analysis which we will examine in chapter 10.

6. Who is to act and use the tools? Five paradigms of organised action

To understand how economic tools for biodiversity are used, how they could be used more, or more effectively for biodiversity, a major step is to clarify who uses them, in the context of what sort of collective action. In more abstract terms, this is the question of agency: who has a capacity to act, and what does that capacity consist in? See for example the call for more use of ETBs in one of the TEEB reports: " *if we fail to account for the value of ecosystems and biodiversity, we will make the wrong choices in responding to these and other challenges.* [...] *understanding and capturing the value of ecosystems can lead to better informed and possibly different decisions*" (ten Brink, 2011, p.3). When "we" are at risk of making the wrong choices, who is "we"? And who exactly is to make better informed decisions? If ETBs can help define goals in terms of biodiversity, whom do we see as the definer of goals? If they are tools for taking action, whom do we see as taking action? If ETBs are ways for increased accountability, whom do we see as being accountable, and to whom?

In this chapter, (1) we shall start by presenting five very different fundamental views on who is to act. Each of them organises in a very different way one possible perspective for analysing and conducting action on collective problems like biodiversity issues; (2) we will then show how a clarification of these diverging underlying views about who is to act can be used to amplify, clarify and organise a great number of questions and issues that we met as we reviewed the difficulties of actually using ETBs, in chapters 1 to 4; (3) then we shall discuss three issues that we think often throw reflection off track and debate on the use of ETBs: failing to realise that the more critical views of ETBs are an integral part of ETB use practice, the ambiguities of the "governance" concept, and the confusion that may result from the fact that we tend to use the same models as approaches for analysis and as blueprints for concrete organisation design.

6.1. Who is "we"? Five paradigms of organised action

Any analysis of biodiversity management action, be it for practical or for scholarly purposes, whatever its discipline or perspective, has to rely on some form of answer to those questions. Such answers can be more or less explicit, but even if they remain implicit, they are of most importance when it comes to drawing from analysis lessons that may be useful for action. It is not enough to state what ought to be done, what tools should be used; one has to consider also who should do it and to examine if they may indeed be able and willing to do it and to use the tools one considers. In former research on the theory of environmental and biodiversity management, we have shown that when one examines many various approaches to environmental management issues, five fundamental answers, five paradigms of collective action become apparent (Mermet *et al.,* 2013). Each one of them brings a very different set of answers to the questions of collective action (Who defines goals? Who takes action? Who is accountable to whom?).

To map out these paradigms, these fundamental concepts of action underlying debates on ETBs use, one has to realise that "society" is fundamentally divided. The utopian view that many seem to hold implicitly, that we 'are all in it together' and thus ought, as it were, to act in unison, is just a utopia, a purely normative concept. It does not lay the ground for a clear account of how decisions are actually made and how management systems function. Neither does it indicate concretely who may start to act and how, if we want to go in the direction of an ideal unified management, while still having to start from reality as it is. In the concrete practice of policy, of resource and biodiversity management, there is no unity of aims, no consensus on responsibility, and there is no such thing as action that would be literally "collective action" if that were to mean that everyone acts together as one, or that we use the same tool at the same time all together. What we do have is partial, contradictory concepts and tools for organised, *joint action of some actors* for purposes that they deem to correspond to *a collective responsibility of all.* To map various fundamental concepts of such collective action beyond the bewildering variety of scales, disciplinary languages and practical controversies, one can examine the controversies, the practical and theoretical discourses on how "society" could or should manage the environment, and look (1) at what is seen to be the main organisational source of the problem and (2) at the views on how the discussion of aims should be organised, and on who should lead collective action. Each of the five paradigms that emerge is like a fundamental cultural perspective on collective responsibility and action, with its likes and dislikes, its heroes and its tools, its buzzwords and its particular feeling of what can be both right, as well as effective in a practical sense.

6.1.1. *The government paradigm*

The government paradigm rests on the conviction that to overcome the innumerable, intense divisions and conflicts in a human group, in a political community, power has to be handed over to a single legitimate actor that will be in charge of collective action. Beyond the array of words and notions – decision-maker, law maker, executive, president, etc. – we will choose the generic term "government" (national or local) to designate an actor that exercises a delegation to lead action on behalf of the collective. Whatever way this delegation is established (democratic or otherwise) it provides the basis for an authority to set goals, identify responsibility, identify and choose preferred ways of action and carry out action on behalf of society.

Important elements of the government paradigm can be underlined in the perspective of clarifying contexts of use of ETBs. First, there is one, clearly identified, decision-maker (or decision-making institution). Second, the other actors are essentially seen as not organised, or not at all on the same level. Approaches that see policy as dialogue between a decision-maker and a non-organised public, or as interaction of a policy-maker with a set of autonomous economic agents, are clearly underpinned by such a government paradigm of collective action. There is thus a clear separation here between the decision-maker on the one hand, and stakeholders on the other. The idea of a cohesive decision-maker making choices on behalf of society is an essential characteristic of this paradigm.

In terms of action and instruments, goals are set by government. The choice of goals is driven essentially by the quest for public good (*l'intérêt general*). Evaluative feedback is pending, through votes, public opinion or other "political pressures", but somehow disconnected (in time, in process, in momentary power asymmetry) from the activity of goal fixing, choosing means and implementing action. Means of action are also set by government. They are driven by its assessment of efficacy, efficiency and feasibility. The use of instruments is straightforward in the "government" perspective: they allow a government to act on its subjects (economic agents, citizens, consumers, etc.). It is because, in the government paradigm, the principal decides and acts separately from the agents (although on their behalf) that the instrumental use of tools appears so straightforward: the one (government) acts on the others (consumers, producers, etc.).

6.1.2. *The coordination paradigm*

In the coordination paradigm, issues in the management of collective affairs – in our case, of biodiversity – are not seen as problems susceptible to solution by a purposeful power, but as resulting from differences and misunderstandings that have to be addressed by the actors themselves. Perspectives embracing coordination as their central paradigm of collective action clearly recognise that the many actors involved do indeed differ on how to act on ecosystems and that they end up in severe problems. But they insist that these same actors potentially have the capacity to solve such problems by themselves if they can only coordinate better. The main obstacles to overcome then are (1) a lack of the sort of communication that can allow them to realise their joint interest in cooperating, or (2) external obstruction to good coordination (for instance, by poor government policies, or by the intrusion of external forces that thwart the actors' efforts for coordinated action).

A reference book from that perspective would be Elinor Ostrom's *Governing the Commons* (1990, see chapter 7, this volume). Ostrom provides examples of how solutions to problems of coordination resulting in resource depletion have been repeatedly achieved in regard to a range of resources. She explores in depth the rationales and conditions for success, and warns how government intervention often makes the problem worse instead of better. Many other approaches espousing the coordination paradigm have been influential in the last three decades, for instance Barbara Gray's "Collaborating" (Gray, 1989), Rhodes (1997), Kooiman (1993) or "*Gestion Patrimoniale*" in France (de Montgolfier *et al.,* 1987). For a discussion of ETB use in the context of coordination-paradigm-based approaches, the following traits will be important to consider.

1. On identifying the operator, two views coexist here: either one considers that the collective of actors "around the table" operates like one decision-maker (and uses decision-making and action tools like one), or one considers that each actor is an operator that uses tools, but in a way that is informed by his participation in a coordinated action (World Bank Operations Evaluation Department, 2002). Both views leave open the question of how precisely this type of decision-making is actually processed. Are decisions reached through negotiation, through cooperation, etc.? Some of the theories discussed in the next chapters will allow differentiated and precise answers to such questions. But whatever the modalities for coordination, two sets of issues will be essential in action (and thus use of tools): (a) the need for abundant information and intense dialogue, because a lack of reciprocal information is seen as a major obstacle to efficiency and (b) the pivotal role of rules, explicit or implicit, in

coordinating behaviour and relations. A major trait of this paradigm is that it abolishes or downplays distinctions between decision-makers, the public and stakeholders: the important point being that decisions are made jointly by entities on an equal footing.

2. In terms of action and instruments, the distinction between goals and means may lose much of its relevance. What is a goal for one may be just a means for another. Actors make decisions jointly but it is not necessary that there be an agreement on goals to agree on a given course of action.

3. In a context of joint decision-making, the concept of "instrument" is not the same as in a context of action of one agent on others, or on a situation. This is well illustrated by the notion of "legal instrument" in international law that designates not a tool, for one operator to act on another, or on a situation, but a set of agreed rules (a convention, a treaty) that allows actors with comparable standing to act on one another in a way that coordinates their actions, as it is typical in Selnes *et al.*'s synthesis (2006).

6.1.3. *The revolution paradigm*

Symmetric to the government paradigm is the revolution paradigm. Here, government, and more widely the existing political regime, is seen not as articulating collective problems and goals, nor as the operator of collective solutions, but as being itself at the centre of the problem, and as promoting fake solutions that are themselves, in effect, part of the problem. Seeking real solutions to collective problems then has to rely on overthrowing the existing political regime and the order (social, economic) it both sustains and is sustained by. In the context of biodiversity this means replacing current orders that put ecological issues at the periphery by a new order that would put it at the centre, or at the foundation of a new political, social, economic, etc., order. For example, some authors who point out that the economic system is entirely embedded in social, and then physical spheres, and should then be submitted to redefined laws, according to the order of biosphere and thermodynamic laws, could be linked to this category, as Passet (1979) or Georgescu-Roegen (1976). Should "we" then overthrow the existing political regime to that effect? For those of "us" who espouse a revolutionary perspective, this is clearly the way to go.

A key concept in the revolution paradigm is that we are all entangled in a system (political, economic and/or cultural) which both destroys nature (and many other human concerns) and hides the process behind a constant barrage of ideological rhetoric. The title of Joel Kovel's book *The Enemy of Nature – The End of Capitalism* *or the End of the World* (2002) summarises in a nutshell one such revolution-oriented diagnostic.

In terms of action, the fundamental operator in the revolution paradigm is the mass of the people, because this is the only entity that can set the goals and norms of legitimate politics and policies, and because it is the only agency, the only force that can overthrow "the system". In practice, the operators are leaders seeking to mobilise against the system. A central issue for action here is to bring large numbers of people to a renewed awareness that would allow them to be conscious of their entanglement and its consequences. This would lead to such a massive shift in values and practice that the "system" would become untenable and the major obstacles to ecologically sound lifestyles would be overcome. On the other side of the action is "the system" (the political regime, dominant forces in the social system, and in many revolutionary writings, the capitalist economic system), that resists change even when such change would be beneficial to all (or most) in terms of ecological sustainability. In practice, analysts grounded in the revolutionary paradigm tend to see the system as deploying a variety of operators, from government services that resist change to corporate lobbies, from media which operate as *de facto* censors against real change to parts of academia that are squarely at the service of ideological discourses that sustain an illegitimate political regime.

The "government", "coordination" and "revolution" paradigms are pure, squarely contrasted models of action for acting on collective problems like biodiversity issues. But contemporary debate and practice is also informed by two more hybrid paradigms, each one combining two of the paradigms we just described.

6.1.4. *The governance paradigm*

The *governance* paradigm is a hybrid between "government" and "coordination". It is usually presented as an opening up, starting from a model where government is the operator and including other actors and stakeholders to participate directly in the making and the implementation of policies. In the context of biodiversity, this opening up is often presented as being made necessary by government appearing to be both over-ambitious and insufficiently effective, or to be too authoritarian and insufficiently open to expectations and expertise from the people (and from local organisations). But many will also concur towards this hybrid model starting from the coordination paradigm, either because they will see that in real contexts, coordination without government interference (or assistance) is rarely possible, or because they consider coordination to require the setting up of institutionalised operators – in essence, new organs of government.

In the governance paradigm, the key to more efficient action is to be found in reinforced cooperation between government and civil society (*i.e.* both the public at large, civil society organisations like NGOS and the private sector). In a governance perspective, government must open to such actors its main decision-making and action processes – the discussion of goals, the allocation of responsibility, the choice and implementation of means. Conversely, the initiatives taken by civil society have to be gradually taken up to become forms of government action (Duran et Thoenig, 1996). For the analysis of uses of ETBs (or any other form of environmental knowledge and tools), this is a particularly complex model for two reasons:

1. the governance paradigm encompasses all the operators and issues of the government and coordination models. There are, however, deep contradictions between a "government" and a "coordination" view of dealing with collective problems. For instance, those who would like to rely more on direct deals between stakeholders often show the negative role of government interference [21]. By merging two contradictory perspectives on collective action, the governance paradigm generates much ambiguity. Since the governance paradigm is so influential in current thinking about ETBs and their use, this ambiguity carries over to ETB debates, both in practice and in theory.

2. approaches based on the government paradigm do not just carry over elements from government and from coordination. They differ from both government and coordination in the way they conceive action and decision. Here *process* – rather than, respectively, optimisation of collective choice or negotiated balance between different individual values and interests – is considered to be the fundamental basis both of decision-making and of action. In practical terms, the main concern is to improve process and interaction rules. In terms of sources of legitimacy, a decision or collective action is seen as sound if it stems from due process. In terms of theories underpinning analysis, analysing decisions and action is equated to analysing the processes of making decisions and of action. In contrast with the government paradigm, there is not one operator, and the problem is not primarily one of making the right decisions but of deciding to follow the right process. In contrast with the coordination paradigm, there is not one set of parties, and joint decision-making does not have the same meaning (process, rather than agreement, makes decisions and shapes action).

[21] The topic will be discussed more in depth in chapter 7.

In short, the governance model for organising collective action relies crucially on mechanisms (*dispositifs*) designed to generate, sustain and manage appropriate processes (*e.g.*: public participation processes in the EU water framework directive (2000/60/EC), or the Kyoto process, or the innumerable processes of participatory planning).

6.1.5. *The minority intervention paradigm*

In the *minority intervention* paradigm, decisive action to take charge of environmental issues is seen as relying on some actors – interest groups, specialist agencies, etc. – acting on other actors to obtain changes in activities and choices (behaviours, development strategies, projects) that damage biodiversity and ecosystem services. Here, responsibility for environmental problems is not assigned to society as a whole, or to "the system" (as in the revolution paradigm), but to some clearly identifiable human causes, to specific powerful actors, activities or sectors. The stake is not to overthrow the system but to transform specific parts of it in ways that lead to more biodiversity and sustainability.

A central concept in this paradigm is that neither government (as in the government paradigm), nor the collective of actors (as in the coordination paradigm) are able by themselves to change course in an amicable manner. Indeed, both government and "the parties around the table" represent the very balance of interests, views and forces that drive development paths that are unsustainable and erode biodiversity. To change course requires the intervention of an agency that does not express but disrupts that balance, so as to disrupt unsustainable courses of collective action.

In that context, the main operator is in a minority position, since they want to change the existing balance of things, the very balance which defines the majority. In the field of biodiversity, it can be an ecological activist group, an international NGO, a government agency specialised in biodiversity that tries to make other government agencies or departments change course on some biodiversity issues, a small organised group of farmers that defend a different model of agricultural development that would be better for biodiversity than the one defended by the mainstream in their profession, etc.

A central point in this paradigm is that the most decisive action relies on agents of change who, at least at the beginning of the action, have less power than those actors and forces (economic, social, political, etc.) who cause loss of biodiversity. Three consequences derive.

The first is that decisive action is strategic in essence. It requires that action be conceived so as to overcome intelligent opposition. For instance, the minority intervention paradigm underlines issues like organised resistance to environmentally-motivated change, which are eluded by the currently dominant views based on government, coordination or governance perspectives. More generally, in this perspective, action for biodiversity is illuminated more by strategic models of action (strategy in terms of management, social movements in terms of politics and sociology, etc.) than by rational or deliberative models of action that underpin the government, coordination and governance models.

Second, in a minority intervention perspective, it is clear that action takes time, for instance the time to build up power for the agent of change, the time of advocacy and to obtain changes in others' thinking and behaviour, the time it takes to transform the system (as opposed for instance to the speed with which the system may be overthrown in a revolution, or with the moderate delay required by the pondering of rational decision-making).

Third, in this model, pluralism takes a more marked form than in the others. It is not seen as a case of actors with various opinions eventually agreeing on a common course of action, but as a joint course of action resulting from interaction and only partial agreements between actors who continue to disagree and pursue their own goals and strategies. For the analysis of biodiversity values, let us underline two consequences here:

First, the minority actor, or coalition, defending a given biodiversity interest (*e.g.*, the tiger, or coral reefs) is only one amongst many other minority interest groups and agencies (*e.g.*, defending the rural poor, or promoting climate scepticism). The question of coalitions and struggles between causes that may or may not converge on a given issue, at a given point, becomes of paramount importance here. The momentous relations between pro-poor and pro-biodiversity actions, between rural development and biodiversity, or between the champions of sustainable forest management and those of forest biodiversity illustrate how useful it may be to adopt a model of collective action that does not espouse *a priori* the principle that all good causes are automatically synergic.

Second, there is no end to pressure, struggle, campaigning, lobbying and other forms of strategising. The vision that these would be only transitional phases leading to agreement on joint action is not warranted: agreement on values is always circumstantial, limited to a set of issues, at a given time, in a given context.

The way the minority actor of change acts can be conceived very differently, both in practice and theory. In Callon's (1986) "sociology of translation" (see chapter 9), he is viewed as an innovator who, as he introduces a new design (technical in his example, but it could also be organisational or financial), tactically manages to realign heterogeneous interests into transformed patterns of relations. In the "advocacy coalition" theory of Sabatier and Jenkins-Smith (1993), public policy changes after advocates of a given policy cause (*e.g.* a biodiversity friendly policy of public forest management) manage, over a couple of decades, to rally a wider and wider coalition advocating that particular course of policy to the point where it is able to change the course of policy in its field of interest. In our own "strategic environmental management analysis" (Mermet, 2011, see chapter 10, this volume), conserving or restoring biodiversity essentially relies on strategic action, using highly diverse strategic means (from buying land to discrediting unsustainable developers, from media campaigns to judicial action) to obtain changes in the actions of private or public actors causing biodiversity loss.

6.1.6 *A view point indicator to sort out perspectives on organised action*

This presentation of five paradigms of collective action can be summed up in table 4, which recapitulates for each who is seen as the main operator of biodiversity management and policy and what interactions are at the centre of such management and policy. The table also lists some of the typical concepts and buzzwords that tend to recur in analyses grounded in the various paradigms.

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Table 4 *Five paradigms of collective action in biodiversity management and decision-making*

Paradigms	Who is the main operator of biodiversity management and policy?	What interactions are at the centre of management intervention and policy?	Typical concepts and buzzwords
Government	A government that has a delegation to act for the collective	Intervention to modify behaviour through various tools and policies	Decision-makers, Official targets, legitimacy, implementation, policy instruments
Coordination	Stakeholders themselves	Coordination and direct collaboration between stakeholders	Actors around the table, co-construction, mediation, collaboration, community
Revolution	Masses and their leaders in opposition to "the system"	Mass action for wholesale systemic change addressing a whole range of societal and environmental issues	Globalisation, commodification, Capitalism, ecological crisis, colonialism, growth as the systemic cause of environmental problems
Governance	A complex set of government and stakeholders	Complex procedures combining public policy and stakeholder participation	Participation, participatory planning, stakeholders involvement, public-private cooperation
Minority action	An actor focused on a specific conservation goal and acting to reach it	Strategic action to obtain changes from specific actors whose activities impact biodiversity	Environmental groups, Activism, Innovators and advocates. Legal or political challenging of decisions

Source: adapted from Mermet et al., *2013.*

The relations between the five paradigms are complex, and we shall discuss them further in the last part of this chapter. But before continuing in that direction, we need to reconnect with the tool-centred perspective we adopted in the first four chapters. We won't proceed to a systematic review, but rather select a few important themes and show how the clarification allowed by the five-paradigms approach opens interesting perspectives for further treatment. We shall start with ESV, and continue with economic tools that put money on the table.

6.2. ESV: who measures value to influence whose decisions, and in what ways?

As we take the measure of how different a perspective each of the five paradigms sheds on action to solve common problems, we start to realise how much clearer the debate on the use of ESV would become if we did not jump as easily and as implicitly from one model of agency to another. In chapter one, we saw that the examination of the difficulties of effectively using ESV revolved around the limitations of the rational decision-making model of policy that underlies the economic theory behind ESV (Laurans and Mermet, 2014).

6.2.1. *More explicit alternatives to the elusive "rational decision-making" model of ESV use*

Much of the literature on valuation is indeed based on the idea of assigning monetary values to biodiversity to allow a decision-maker to make choices, to weigh between various interests through an arbitrage based on common interest, represented for instance by the maximisation of well-being through quantitative indexes that aggregate biodiversity and other values. However, despite the attention the principle has received over the last fifty years, and despite the available toolbox, such uses remain disappointingly scarce (Pearce, 1998; Fisher *et al.,* 2008; Liu *et al.,* 2010). Our own research has demonstrated that cases of actual use of cost-benefit analysis to feed decision-making choices appear to be very rare in the literature (Laurans *et al.,* 2013). As soon as one examines more closely the dynamics of the use of ESV in the field, one finds out that real decision-making about biodiversity issues is usually too political to fall into the pattern of rational decision-making by a decision-maker. This explains why, of the various possible uses of ESV, the ones that are most elusive in actual practice are the ones that embody the rational model of decision-making – in particular, decision-making based on cost-analysis studies, or "technical" use of ESV to establish levels of payments for PES.

Even when economic studies are used by government for making decisions, the process (and the content) is quite politically laden. Consider for instance the French government's assessments of the future impacts on the economy of recent environmental laws. Following a major national environmental policy conference (the *Grenelle de l'environnement*, 2007), a set of ambitious measures were worked into law over the following years. But as they became law, and thus had tangible impacts, the political commitments became contested. To counter this, the Ministry of the environment commissioned an appraisal of the positive impacts of the environmental laws by the Boston Consulting Group. It produced impressive figures related to the number of jobs that the works and the regulations resulting from the new environmental requirements would induce (The Boston Consulting Group, 2009). But it was contradicted a few months later by a study made and issued by the Treasury Department (Birard *et al.,* 2010), which assessed the detrimental costs, for public finances, from the additional spending resulting from those environmental laws. One cannot resist seeing here, rather than a weighing of costs and benefits from a unique point of view, supposedly meant to achieve an optimal calculation, a traditional struggle for influence between "spending" ministries and "financing" ministries, the result of which is determined by negotiation rather than calculation.

Such back and forth of advocacy and justification that underlies political decision-making is in fact the natural terrain for the use of economic studies and particularly of ESV in actual practice. Its stakes and dynamics are quite different from rational arbitrage based on a cost-advantage study.

6.2.2. *ESV as advocacy*

This is why the use of ESV as a tool for advocacy is so pervasive. It is implicit in the position of many or most of the environmental economists promoting biodiversity valuation. Through their various methods, they are trying to demonstrate that some values of biodiversity are overlooked, whereas they ought to be taken into account in public decisions. In effect, the work of such economists is a plea directed at other actors in society to convince them, based on evidence from economics, of taking biodiversity into account more than they do now. This intention and position are exemplified in a striking way by the TEEB report. In its page called "some numbers" of the synthesis report, high figures are pulled together, obviously to sustain arguments like the one that follows 2 pages later: "Finally, the failure to account for the full economic values of ecosystems and biodiversity has been a significant factor in their [ecosystem services] continuing loss and degradation" (Sukdhev *et al.,* 2010, p. 9). Far from being an exception, this advocacy component of TEEB reflects a founding dimension of environmental economics at all scales.

Now whom are advocate-economists trying to influence through ESV ? Is it the decision-makers themselves? If so, are they trying to influence arbitrages by governments between biodiversity and other concerns? Or are they fuelling the complex public debates of governance in the hope of influencing their outcomes? Are they providing arguments to actors focusing on defending the cause of biodiversity? But mostly, the very principle of being engaged in advocacy puts them in the situation of "minority actors" speaking to government or to other actors. One of the signs of such a position is the feeling of speaking to a deaf ear felt by many environmental economists. As one of our interviewees put it: "In Costa Rica, many tried to promote the environmental cause since long, but it didn't work well, at first. Then we told the Ministry of Finance what it cost to his country's GDP… Even if the figures were inaccurate, it contributed to discovering [a] new system of values". This is inherent to all minority intervention situations. If the majority of actors were not resisting, somehow, taking into account some important values of biodiversity, there would be no need for advocacy in the first place: all the economist would have to do is watch as economic agents adjust their behaviours to those values as they get the appropriate information on them, or just provide the relevant information to governments, who would then, without fuss, accordingly modify their arbitrage on the issues.

6.2.3. *For decisions based on deliberation, justification is a fundamental and legitimate use of ESV*

One other use of ESV that is often encountered in the field is for justification of decisions that have already been taken on political grounds. Consider for instance the following example, taken from our own experience, of studies to assess the economic consequences of river basin management plans, in 1996 and 1997. Analysis of the decision contexts showed that conducting economic valuations was not in fact meant to decide based on cost-benefits ratios, but was intended to help specific professional groups (industrials, large municipalities) to assess and demonstrate the economic efforts the management plan would require from them (Laurans *et al.,* 2011). Likewise, the (Green Party) elected representative from the North-Pas-de-Calais Region, steering the economic valuation committee of the Artois-Picardie River Basin management plan, declared that he was not opposed to monetisation of environmental benefits, and wanted some contingent valuation and willingness-to-pay figures, because it would provide him an appraisal of the possible consent of his constituents for this program of measures (Laurans and Dubien, 1996). In such examples, even though valuations are commissioned by governmental entities, they do not appear as a tool for an optimisation calculation, but rather a means to communicate, advertise, argue, and negotiate.

If we had to choose only one area of investigation to improve the use and impact of ESV, it would probably be the in-depth exploration of the dynamics of such communication, advertisement, argumentation and negotiation. This is all the more relevant as the current prevalence of a "governance" view of dealing with environmental issues induces a proliferation of complex and ambiguous participatory decision-making processes involving all sorts of operators and stakeholders, from government to firms, from NGOs and universities to the lay public. In such procedures, justifying one's point of view, one's plans and decisions, is essential. This may well be one of the reasons for the rapidly increasing demand for evaluation in general and for economic evaluation in particular. Our review of uses of economic evaluation of biodiversity shows many examples where such evaluations are part of the procedural steps requested in the establishment of action plans, the European water framework directive being a very good example, on a large scale. The current efforts to upscale the use of ESV seem to us to be part of that particular dynamic of environmental governance.

A deeper study of ESV use for justification will be both particularly useful and somewhat challenging. Indeed, the use of ESV for justification is often identified in the literature, but tends to be treated in a derogatory way, as if it were a let-down from what would be serious use, *i.e.* calculation to prepare a rational arbitrage. However, if one takes seriously the idea that decisions (i) should be made with a large input from the public and from stakeholders, and (ii) should follow in-depth deliberation, then justification is to be considered the very centre of decision-making, rather than ad hoc after the fact.

To sum-up, we need to study the use of ESV not only in the perspective of a rational decision-making government, but also of clearly stated alternative models of agency.

6.3. Economic tools based on payments, markets and property rights

As we turn from valuation to economic tools that actually put cash on the table – through payments, markets, property rights – we also find that the difficulties and debates we reviewed in chapters 2 to 4 revolve around the fact that the "coordination" model of agency at the same time (a) dominates the discourse on market instruments and (b) is unable to account for the actual dynamics of biodiversity management situations in ways that would be sufficient to guide practical use of the instruments.

6.3.1. *Pure coordination as the elusive utopia of tools based on actual economic transactions*

The most telling example is the tension around the archetypal definition of payment for ecosystem services as "a *voluntary* transaction where (b) a *well-defined* environmental service or a land-use likely to secure that service (c) is being "bought" by a (minimum one) service buyer (d) from a (minimum one) service *provider* (e) if and only if the service provider secures service provision (*conditionality*)". Whereas this definition clearly underlies the fundamental principle of coordination – a direct deal between agents set on an equal footing, without interference (*e.g.* from government) – we have shown in chapter two that to understand the use of PES in real management situations, one has to take into account other patterns of agency like, for instance, special-interest groups or government acting as buyers, or government intervening to help (or to block) direct-coordination deals. The same remarks hold for tools based on land-rights – their effectiveness lies not so much in autonomous market dynamics of land property as it has to rely on (and later, facilitate) complex multi-actor deals – as well as for biodiversity-banking (where trading compensation is not an alternative to but a part of heavy government intervention). For all three types of tools we also found that intermediaries (consultants, experts, trust-funds, brokers, etc.) played a decisive role in the design and implementation of economic tools on the ground. The conclusion we derive from this is not a call to dismiss the elementary agency model of direct coordination through market, but to embed it (a) into a fuller picture of the coordination mechanisms at play and (b) into a wider picture of patterns of agency that are not centring on coordination.

In French, the word *marché* expresses nicely the ambiguities of the market's place in the fuller picture of coordination: it means both a "market", and a "deal". This encapsulates a conclusion that emerges from reviews of the actual use of monetary instruments in the field of biodiversity: there are very few markets (in the form of organised encounters and monetary exchanges of potentially multiple buyers and sellers of the same service), and many deals, *i.e.* multi-dimensional negotiations and agreements of which the transfer of funds is one item, the meaning of which can be assessed only in the context of the package deal it belongs to. One of the main perspectives to improve our (analytical and practical) grasp of the use of ETBs for coordinating actors with different needs, values and means would be to invest in the analysis of the negotiations of such deals, including their monetary components but in no way presupposing a centrality that may not reflect the real conditions of actual management situations.

Beyond coordination, each of the other paradigms of organised action sheds a different light on the meaning and practice of ETB use.

6.3.2. *Tools that take on a different meaning from a minority action, government, and governance perspectives*

Starting with minority action, ETBs are essentially part of the various tools that actors promoting biodiversity use to try and change the behaviour and projects of others. In mechanisms like PES, or buying conservation or restoration, the payer or buyer is very often the one actor – or the coalition of actors – who wants more biodiversity, paying to other actors who would be content with less: a water agency pays farmers for reducing fertilisers, the Nature Conservancy buys land that would otherwise be developed, an NGO compensates sheep farmers for the damage done by bears or other endangered predators. Whatever sort of monetary instruments are considered here, they are to be taken not so much as tools in the economic toolbox than as part of the acting-in-favour-of-biodiversity toolbox. For instance, seen from the point of view of those actors who act in favour of tropical forests, REDD funding is analysed and used as a (powerful and dangerous) tool in their limited arsenal, alongside scientific expertise, activist networks, legal instruments for forest protection, etc. Since a chronic problem with minority intervention is that the toolbox is always felt to be under-equipped, there is widespread eagerness from promoters of biodiversity to participate in the development of all new tools, including ETBs. The pink book of the Global Canopy Program (Parker and Cranford, 2010) is a good example of enthusiastic shopping in the ETB section of the economic hardware store to strengthen the toolbox of those defending tropical forests.

If we turn to the government paradigm, ETBs are to be understood as part of the panoply for government intervention. Although it lies in deep contradiction with the coordination foundations of tools based on markets or property rights, this alternative identity of ETBs is certainly very prominent in practice, as illustrated by subsidies for environmentally-friendly farming, the buying of land and easements by public authorities or the pivotal role of government in compensation and biodiversity banking. In this perspective, ETBs are a part of (not an alternative to) the government intervention toolbox alongside regulatory, administrative or discursive tools in combination with which they are used.

Finally, it strikes us that the rise of the governance paradigm, by encouraging public-private partnerships, creates a context that is very favourable to the use of monetary tools such as PES, or the buying and leasing of ecologically sensitive land. First these tools are extensively used both by private and by public operators. The limits between public and private action has little relevance for them, by contrast to regulatory instruments, for instance. Then, as we have also seen in our examination of biodiversity banking, the actual implementation of ETBs often relies on the constant interaction of government authorities and private operators. The use of most monetary tools for biodiversity is characterised by hybrid, fuzzy mechanisms, governed by hybrid panels of heterogeneous operators. These are precisely the sorts of mechanisms that are supported by a "governance" view of action, whereas other models of action find them much more problematic, through the lack of clarity of who is in charge of what (seen from a "government" perspective) or because of the opportunities for manipulation of decision-making processes to the advantage of private interests or of the established political regime (seen from a "revolutionary" perspective).

6.4. Clarifying alternative models of agency is a continuing challenge

To sum up, as we examine the state and issues of ETB use described in chapters one to four in the light of the five paradigms of organised action, we can derive the following two conclusions: (1) the constant (largely implicit) use of alternative, partly overlapping and partly contradictory models of agency is a major factor in the difficulties met in the diagnostic of ETB use problems and of the debates surrounding the development of ETBs; (2) clearly laying down such models of agency provides a fulcrum to leverage much-needed efforts for clarification, both in the general debates and in field situations of ETB use. Of course, beyond the didactic simplicity of the five paradigms it takes a real effort to untangle the complex and ambiguous webs of agency in real-life biodiversity management situations. We shall devote the rest of the chapter to three different issues that often come up and block the road to clarifying questions of agency in the use of ETBs: (1) the role of the revolutionary paradigm in the ETB debate, (2) the problematic ambiguities involved in the concept of "governance", and (3) problems that arise from the shifts through which the same words are used in the debate to refer to practical mechanisms and to analytical interpretations of situations.

6.4.1. *Why the wholesale critique of economic tools should be a part of the ETB-use debate*

Whoever starts considering economic tools for biodiversity is, before long, confronted with intense reactions against the very principle of such tools. Negative reactions are frequent from many people involved in biodiversity issues. On the more academic side, there is an important current of thought that supports wholesale rejection of all economic tools for biodiversity. Its underlying theory of action is clearly – in various forms of course – the revolution paradigm, that is, the idea that it is the system itself (especially the government-supported economic system) that is at the roots of the most serious ecological problems. If this is so, then supporting whatever economic tools that strengthen the system is counterproductive for biodiversity.

Such positions that are critical overall hold a paradoxical position in debates about ETBs. They tend to be downplayed in the ETB literature, since so much of it is based either in journals in economics or in grey literature about ETBs, neither of the two being particularly interested in views that oppose economic tools in general. But on the other hand the body of literature that criticises the very principle of bringing the treatment of biodiversity issues onto economic grounds is considerable and hard to ignore. See for example Clark *et al.* (2000), who suggest nonsense and value incompatibilities when people answer to willingness-to-pay questions. This literature is to be found in journals centring on philosophy or on political science and more or less disconnected from discussions amongst specialists of ETBs. We shall not review it here, but as will become apparent, it is important to introduce some of the main arguments involved in our analysis.

We will do so based on John O'Neill's book *Markets, Deliberation and Environment* (2007), which introduces, in a compelling way, many of the most important points made by the more radical critical view of ETBs.

Overall, his position is unequivocally against the development of ETBs in all forms and all contexts. The central problem upon which he founds his critique is that the use of ETBs leads to, or amounts to, commodification of nature and ecosystem services, and thus makes life of the people and communities poorer, by reducing it more and more to an impoverished system of economic exploitation (of man and of the earth).

For O'Neill, there is no difference to be made between those tools that mobilise actual money (like PES), and those that just provide monetary valuation. While capture by the capitalist system of the former is obvious, the latter also represent such capture, only in another form: the gradual reduction of thinking and debate to economic considerations and an economic language impoverishing social life and political debate and strengthening the hold of capitalism on all aspects of life. This point is important to note in the context of the ETB project: it binds together (in its specific perspective) the two classes of economic tools that we have decided to consider jointly.

For O'Neill, the use of ETBs as seen by radical critiques can be summarised thus: they are used by agents of the system to colonise domains (geographic, ecological, thematic, institutional, etc.) where lively communities are still in functional, social and ecological relationships. So the use of such economic tools reinforces what is already the root-cause of the social and ecological problems and thus only makes them worse. For operators who want to improve social well-being and environmental health, the only good use of ETBs would then be to refuse them upfront.

One might conclude that such views should have little or no place in pragmatic discussions between people who – as we are doing with this book – are trying to examine how and to what extent ETBs could be mobilised more actively and more effectively to conserve and restore biodiversity. Each camp, as it were, would just go on with its business, the ones trying to make economic tools work and the others explaining why they do more harm than good anyway. In our view, we should go the opposite way and make a lively debate between promoters and critiques of ETBs an integral part of a really pragmatic approach to ETBs. Two reasons point in that direction.

First, as we have argued at length already, using ETBs makes sense only in the context of wider strategic action in favour of biodiversity. And balancing between rejection and use of a given set of tools is an integral part of strategic debate about strategic action for biodiversity. For O'Neill, pressure towards the use of ETBs puts people who are dominated in the system – and advocates of a more ecological way of life – in a situation of dilemma. He sees them as being cornered in a position where they have to choose between using ETBs to obtain better terms, at the cost of reinforcing a system that works against them and against sustainability, or refusing to use ETBs and accept worse terms for themselves within the system than they would have, had they used ETBs. In actual use, the question "who does the use of a particular tool reinforce?" is indeed a crucial one, one that often tends to be eluded, and that critical approaches usefully keep alive. The more systemic consequences of using ETBs are an integral part of the choices we make in using them (or not). And the possibility that one may choose not to use ETBs for strategic reasons although one accepts their principles, or conversely to use them sometimes against one's general principles, are equally highly practical possibilities that are enlightened by critical discussions.

A second reason for being attentive to the more critical views about ETBs is that quite often they point to issues that may be taken not so much as reasons for *a priori* rejection, but as important points to consider for avoiding pitfalls in practical use. The critique by O'Neill of biodiversity banking, for instance, points to issues (the limits of establishing equivalences, the inevitable infringement of the sense of place of inhabitants inherent in all land planning tools, including biodiversity offsets, etc.) that are at the very heart of the practical challenges one meets once one has in fact committed to implementing biodiversity banking projects. We are struck to see with what intent and care such issues are often discussed by the practitioners and promoters of biodiversity banking themselves. There is little doubt that the strong critical pressure on the very principle of compensation and biodiversity banking is a major driving force behind the improvement of how carefully the schemes are designed and implemented, and thus, that critics contribute to sharper thinking and better practice in the field.

A core argument of our own analysis is that economic and other dimensions of biodiversity management situations simply cannot, and should not, be isolated from one another. The use of any tools for biodiversity, economic or not, is embedded in social and managerial strategies and interactions. These strategies and interactions, not this or that tool *per se,* are decisive. Trying to exclude economics from the politics of biodiversity is equally unrealistic and undesirable as trying to exclude politics from the economics of biodiversity. The politics of ETBs and the practical use of ETBs are inseparable. As we concluded in the previous chapter, the practical issues of acting for biodiversity here and now and the perspectives of deeper social change with biodiversity issues should be dealt with jointly. The critiques of ETBs in general are part of the scope of the practical use of particular ETBs.

6.4.2. *The ambiguities of "governance"*

Another strong presence in general debates on ETBs and their contexts of use – from quite different quarters than the critiques we just discussed – is the notion of governance. It is used extensively, but with meanings sometimes so broad that it may become a source of confusion. Especially since we also used the word ourselves in our review of paradigms of organised action, we need a deeper discussion of "governance" as a concept. We expect that discussion to help limit the pervasiveness the concept has acquired and put this concept back in its place and so fully realise again the parallel relevance and massive presence of the other four paradigms of action. An in-depth discussion of "governance" as a concept and model for understanding environmental management may start from the fact that the word "governance" is used in the context of "environmental governance" with two different meanings.

Many authors and actors use the concept in a focused and circumscribed way to describe a collective action model where state and other actors work together, as in Floranoy's definition [22]: "Environmental governance can be defined [...] as multi-level interactions [...] among, but not limited to, three main actors, i.e., state, market, and civil society, which interact with one another [...] in formulating and implementing policies in response to environment-related demands [...]". This is the meaning we use here, to contrast the "governance paradigm" with other models of decision and action (government, coordination, etc.).

But increasingly, "governance" is used as an all-encompassing notion of governance as "the act of governing – it relates to decisions that define expectations, grant power, or verify performance [23]". With that definition, governance encompasses the entire field of collective action on environmental issues. Lemos and Agrawal (2006) then propose the term "hybrid governance" for governance in the more specific sense we have used here.

This is a conceptual move already termed the metonymical hustle. *i.e.* what was a part of the picture (in the case of "governance", interaction between State, market and civil society) becomes the whole of the action, and what was a wider concept (public action to deal with public problems), is demoted to become just one aspect of the newly proclaimed whole picture (Mermet, 2009). Such moves are problematic in general because they fuel a gradual shift of meaning that confuses serious, long-view debate on management and policy. In the particular instance of "governance", we consider using the notion to designate the whole picture of collective action to be quite problematic. It covertly generalises the particular, focused and partial perspective of governance as one aspect of collective action. As it does so, it promotes a pervasive sense that multi-institution, multi-level dialogue is not just one aspect, but the very centre of environmental decision-making and thus of the use of ETBs. The result is precisely the confusing picture of ETBs being used in an endless suite of labyrinthic and heterogeneous processes that currently makes more difficult the clearer analysis of pointed use in given practical situations.

A deeper look at the use of "governance" as a concept is all the more urgent in the case of economic tools since the very principles of governance – a constant influence of stakeholders on government, a multiplication of public-private partnerships that constantly blur the lines between government and the market – are in considerable contradiction with the fundaments of economic theory, that look for market mechanisms unimpeded by constant political dealing and for governments to be able

[22] http://ecogov.blogspot.com/2007/04/

[23] http://en.wikipedia.org/wiki/governance, accessed 23.3.11.

to send price signals without being strategically manipulated by economic agents. Well... manipulation of principals by agents is precisely the yardstick of good governance, for instance through successful public participation. There is a stark contrast between the principle of a rational decision-maker using information from economists to make an optimal choice, and the multiple steps of complex and fuzzy procedures of contemporary environmental governance – *i.e.* between a central tenet of environmental economics, and the currently prevailing "governance" view of how we ought to decide about environmental matters.

So the rise of economic evaluation and the mobilisation of monetary tools in the context of ever more hybrid decision-making procedure and management systems generates a contradiction that is both deep and, we think, not clearly perceived by the actors and by the ETB literature. This may be an important element of the malaise that can be felt currently, as we oscillate on the brink of embarking on large-scale use of monetary tools for biodiversity. On a relatively local scale, or as long as payments remain marginal, hybrid arrangements – "deals" as we discussed in chapter 2 – are manageable because they rely on already existing political, managerial and economic systems. But as the scale of use increases, there is a more and more tangible possibility that such tools may transform large-scale balances of power and wealth, in ways that can be accounted for neither by clear government processes, nor by straightforward market mechanisms. The anxiousness and fever raised in the operators and stakeholders of forest environmental issues, and of the forest sector, by the prospects of REDD+ channelling vast amounts of money into social, political and economic systems that may be transformed in major and unpredictable ways is a striking example here.

6.4.3. *Conceptual frameworks for analysis, or models for practical organisation?*

The hybrid character of governance as a model of organised action and the current overwhelming influence of that model, point towards the striking heterogeneity, complexity and instability of the current context of ETB use and of the debate on ETBs. In our effort to make more explicit the different underlying models of agency which are at play in the diagnostic of ETB use situations and in ETB-related debates, we need to introduce here one more step of clarification. Models of agency such as the five paradigms we use here can be used (a) as conceptual models to guide analysis of real-life management processes, or (b) as blueprints for practical organisation, or for both purposes concurrently.

Let us give an example. If one considers that participation of stakeholders in making decisions about biodiversity is essential, and thus, that stakeholders' input into ESV is necessary for the relevance and legitimacy – and thus, for the useful utilisation – of ESV, this concern can be met through two very different routes. The first and most obvious one is to conclude that one must organise more stakeholder participation and concretely experiment with putting stakeholders around the table in ESV pilot projects. The second, maybe less obvious one, is to examine in-depth stakeholders' input into ESV, whatever its claims about being participatory. One will find out that there is always stakeholder participation and input – for instance, through the use of data produced by stakeholding organisations, through the influence of such organisations in the governance and funding of valuation projects, through political and media pressure on the work and result dissemination of valuators, through the stakeholding academic and professional affiliations and preferences of the valuators themselves. This short list of examples suggests that such input, far from being marginal, may influence valuation in a deep and pervasive way, even though it may be overlooked and sometimes denied, *e.g.* through claims of the valuator's neutrality.

The same two sides apply to all models of organised action. A focus on governance may mean a call to organising more concrete interaction between government and civil society, or a call to focus analysis of policy and management more on the processes of interaction between government and civil society, private and public sectors, various institutional levels, etc. A focus on coordination may be interpreted as the need of actually putting people around the proverbial table of collaborative management of biodiversity, or it may mean examining how people coordinate themselves – including in implicit, discrete or even covert ways – in the actual processes through which they manage biodiversity. The same applies of course, mutatis mutandis, to the government, minority action and revolution paradigms of organised action.

As we use the five paradigms of organised action to help us decipher the complexity of debates about ETBs and about their use, it is essential to stay alert to their dual meaning as both analytical concepts and organisational blueprints. We have to pay attention to the ways they are used sometimes to promote putting in place practical projects or organisations (create a new government agency, a stakeholders' round-table, a boycott of decision-making procedures, etc.), sometimes to a call for a different interpretation and analysis of how policy and management work and what is important in them, and sometimes for an (often not very explicit) combination of both. The crisscross of diverse interpretations and practical propositions testifies to the current vitality of the ETB field. It can be even more fruitful if there is not too much confusion of proposals and of perspectives, if we engage seriously in each of the very different avenues of research and experimentation they open and if, accordingly, we keep to a reasonable level controversies fuelled by the misunderstanding, competition or hostility that are inevitable, to a degree, between the various perspectives at play.

To sum up, we propose a second table of the five paradigms, the columns of which summarise respectively (a) the sort of analysis the paradigm calls for, (b) the sort of practical action it provides the rationale for, (c) the typical agent expected to carry out action, (d) the conditions that give authors and analysts who adopt a given paradigm a feeling of achievement, (e) the blind spots of that paradigm (cf. Table 5).

Table 5 *Five Paradigms for analysis and for practical organisation*

Paradigm	Focuses on analysis of…	Promotes practically the…	Puts at centre of action the…	Analyst's feeling of a work well-done based on…	May suffer as a blind spot the…
Government	choices made on behalf of society	determination of best policies and design of effective instruments	decision-maker, policy-maker, manager	clear official goals and well-defined instruments	politics, irreducible diversity of views
Coordination	negotiations and agreements between stakeholders	collaboration mechanisms & institutions	stakeholder, negotiator, mediator	stakeholders at last around the table	weight of government, stakeholders out of reach of direct negotiation
Revolution	fundamental contradictions of the political and economic system	critique of and struggle against the system	critic, activist	compelling argument shows there is no credible solution within the system	practical action to improve the current situation
Governance	processes of interaction involved in policy-making and management	multi-party, multi-stakes, multi-level deliberative procedures	processes & mechanisms, their participants, designers, facilitators, guarantors	sophisticated, inclusive procedures for participation and deliberation are promoted	substantial issues, political struggles
Minority action	targeted strategies to obtain specific changes	advocacy and mobilisation in favour of specific biodiversity outcomes	advocate, professional or official "on a mission"	specific biodiversity issue is making good progress	shared goodwill, cooperation

Source: authors.

Conclusion

Debates about ETB use can leave the uncomfortable impression of being caught between two evils: on the one hand, the simplistic views of action underlying the textbook principles of ETBs, on the other hand the confusing message that it is all very political and depends on complex processes and situations.

On the first side of the alternative, the five-paradigm approaches point up that indeed, the economic theory roots of ETBs limit themselves to two models of action only: market-coordination and government intervention. These two models, however, are not in essence simplistic. There is no reason to reduce them to textbook caricatures. The pragmatics and politics of government intervention, the pragmatics and politics of direct coordination between stakeholders are legitimate fields of investigation and practice by themselves. As long as one is aware that they can't cover the whole story, they deserve serious investigation and each one opens avenues for action with ETBs and other tools.

On the other side of the alternative, the approach proposed in this chapter lays down clear landmarks to navigate the plurality of views, especially of alternatives to the market-coordination and government-intervention views of biodiversity policy and ETB use.

There is not on the one hand a simplistic view and on the other, one complex alternative. There are several different possible perspectives on who can and should act on biodiversity issues, on who can and should use ETBs. Each perspective is powerful and brings a large, specific potential for grounding both analysis and practice in specific ways.

To use them in a relevant way, a few principles have to be kept in mind. There is no model of action that would include and subsume the others. Any perspective on, and for, action is inevitably partial. One has to choose. As summed up in table 5, lucid consideration of the context – especially of what kind of outcomes we are after, and what role we play in the situation as actors or analysts – can help choose the most relevant perspective. There is also no perspective on agency in the use of ETBs without serious blind spots. Knowing where they are helps greatly to draw up more relevant analyses and designs. Conversely, ignoring them just fuels confusion in debates about ETBs.

7. Getting the institutions right? ETBs in the light of common-pool resources theory

As we now turn to examining four theories and frameworks that can help articulate economic tools with social, political and cultural contexts, we would like to start with common-pool resources theory because it is by far the best known of the four, and because it is the one closest to the perspective of biodiversity economics. In essence, it proposes a widening of the economic perspective. As write Ostrom, Gardner and Walker (1994, p. 28), "Markets, hierarchies, and collective action are sometimes presented as fundamentally different "pure types" of situations. Not only are these situations perceived to be different but each is presumed to require its own language and explanatory theory, *e.g.* scholars who attempt to explain behaviour within markets may rely exclusively on neoclassical microeconomic theory. [...] Such a view precludes the development and use of a more general explanatory framework that, together with its constituent theories, could help analysts make institutional comparisons and evaluations." Such a call for integration of market tools in wider action contexts, and the proposal to provide a framework and theoretical resources to do so, fully concurs with the agenda we have set out in this book. A second reason to examine CPR theory is that it is now familiar and influential in the practical and academic scenes where biodiversity in general, and the social and political issues of ETBs in particular, are experimented with and discussed. True, this connection occurs mostly because of the overlap between the field of biodiversity and ecosystem services on the one hand, and the field of rural renewable resources management (water, forest, fishery and the like) on the other. As we will discuss later in this section, CPR theory addresses mostly the latter set of issues – *i.e.* rural resources management.

Overall, it seems well worth examining the potential of CPR theory to provide a structured, theoretically explicit and consistent approach for analysing the use of ETBs and its contexts. We shall start by a brief presentation of the CPR perspective and approaches. We will then focus more precisely on the Institutional Analysis and Development framework which underpins CPR theory, and we will show how it frames the analysis of action situations in terms that create a deep compatibility between economics and social science aspects of resource management situations. Based on this clarification, it will then be time to examine the particular meaning ETBs take when looked at from a CPR perspective, *i.e.* ways to add new rules to existing institutional arrangements concerning the management of an ecosystem service. We will also see how the abundant resources provided by CPR scholarship for diagnosing resource management situations can be used for analysing situations and contexts for the use of ETBs. Finally, we shall look at some limits of using a CPR perspective in dealing with biodiversity issues.

7.1. The common-pool resources perspective

Since its first book-length and strikingly clear introduction in Elinor Ostrom's 1991 book on Governing the Commons, common-pool resources theory has now become familiar to most readers interested in environmental issues. Let us just introduce some central elements that are particularly relevant to our topic.

7.1.1. *Extricating ourselves from the dilemmas of free access*

Leaving aside applications outside the environment and resources field, CPR theory addresses first and foremost the problems that arise when multiple users of a renewable natural resource risk depleting it because (a) each one of them has an incentive to appropriate more of the resource before the others do, and (b) there is no effective system to limit extraction (what is called a "free access" situation). Some of the most serious processes damaging biodiversity (overfishing, part of deforestation, drying up of water ecosystems and wetlands, species extinctions from overhunting, etc.) fall into this category.

CPR theory originates from a reaction against the claim that there would be only two solutions to such problems: setting up a private property system (so that each owner takes care of balancing his extraction with his now private property resource-base), or state-imposed regulations to limit extraction by users. The central point of CPR theory is that there is a third way out: users can, in many cases, extricate themselves from the dilemmas between individual and collective interest that lead to over-extraction of resources. To do so they have to create institutions, that is, in the CPR theory perspective, systems of rules that allow a coordination of their behaviour in ways that maintain a sustainable level of resource extraction at the collective level. From a CPR perspective, the crux of resolving environmental problems is in "getting the institutions right" (Ostrom, 1991, p. 14).

Of major importance, however, is the fact that, in Ostrom's words, "the capacity of individuals to extricate themselves from various types of dilemma situations varies from situation to situation." (*ibid*, p14) Details in the ecological or geographic situations, in the culture, in the social system, in legal or administrative rule and practice, in technology, etc. can make great differences in the problem to be solved and the kind of institutional solutions that may work or not. As a result, there is no general form of solution, no institution that is right in general, no panacea (Ostrom, 2007). Consistently with this point, much of the effort of CPR scholarship has gone into documenting and analysing a large number of cases where people have succeeded or failed at designing and implementing a system for the sustainable use of resources shared in common. Such studies are now very numerous in the CPR literature. The site of the International Association for the Study of the Commons, or the Commons Digital Library (at the Universtiy of Indiana) makes it easy to access that wealth of field-research examples.

In the eyes of CPR scholars and practitioners – and we certainly concur with them on this point – there is simply no way around producing a specific diagnostic for each situation where one wants to intervene. However, CPR scholars have endeavoured to extract from their large base of cases a series of factors that can make it easier or more difficult for resource users to organise themselves and "get the institutions right". Table 6 presents the list of such "design principles" in Ostrom's 1991 version.

Table 6 *Design principles for successful common-pool resources management*

1.	*Clearly defined boundaries (of the ecological system at stake, and of the set of appropriators).*
2.	*Congruence between resource use and maintenance rules and local conditions.*
3.	*Collective-choice arrangements whereby most individuals involved can participate in defining the rules.*
4.	*Active monitoring of the resource, conducted in a way which makes them answerable to resource users.*
5.	*Graduated sanctions in case of a user violating the rules.*
6.	*Easily available conflict-resolution mechanisms.*
7.	*Sufficient recognition of the right to organise (i.e. government does not interfere in local self-management in ways that defeat the capacity of users to regulate by themselves the use of the resource.*
8.	*Additionally, for CPRs that are parts of larger systems, all these conditions have to be met at multiple levels of nested management systems.*

Source: Ostrom, 1991.

Many variants of such a list of conditions are found throughout the CPR literature. Some are shorter, some more detailed, but the main elements as presented here are essentially stable and present sufficient common ground for the purpose of our discussion in this section.

From this list of design principles, it is clear that CPR theory shares many tenets with the sorts of approaches that today tend to dominate the intellectual landscape in biodiversity management. Much of the literature in the field tend to promote local management over more centralised systems (as reflected in CPR design principles 1, 2, 3 and 7). It insists on cooperation and conflict resolution, as reflected by design principle 6 and as inherent in a form of management that rests on users reaching agreement between themselves. It sees rules and their implementation as playing a major role (as reflected by principles 5 and 6). These traits of the CPR perspective echo respectively (a) the decentralising and localist, (b) the cooperation-oriented and (c) the institution-centred trends that are very influential in the environmental management literature of the last two decades.

7.1.2. *The IAD framework: a deeply seated intersection between economics and social sciences*

However there are much deeper roots to CPR theory than such general pragmatic design principles – a squarely defined foundational framework, a well-circumscribed set of theoretical resources and a consistent methodological repertoire. These are highly relevant as our purpose involves identifying theoretical resources with reach and substance for in-depth analysis of ETB use situations. Let us then present the CPR foundational framework and then move on to the methodologies.

In her most systematic presentations of CPR theory, Ostrom insists that the whole approach is based on the Institutional Analysis and Development (IAD) framework. In her own words, the framework provides the "general scaffolding that supports [CPR scholars'] enquiries" (Ostrom *et al.,* 1994, p.27). It makes explicit the underlying paradigm of which the various theoretical tools mobilised by CPR research are "variants" (*ibid*).

The IAD framework sees all situations as "being composed of the same set of elements". Resource-use situations in the case of CPR research – biodiversity management situations in our case – are to be considered as "action arenas". All action arenas are best analysed by identifying seven clusters of variables. The framework is summarised in figure 8.

Figure 8 *The main concepts in the IAD framework*

An action area is composed of

An Action Situation involving

Participants in
positions who must decide among diverse
actions in light of the
Information they possess about how actions are
linked to potential
Outcomes and the
Costs and Benefits assigned to actions and outcomes

Actors, the participants in action Situations who have

Preferences,
information-processing capabilities,
Selection criteria, and
Resources

Source: Ostrom et al., *1994.*

This framework is often overlooked when CPR is discussed in a context of application to biodiversity and resource management, as most commentators focus more pragmatically on the design principles that facilitate cooperation and on case studies. It is difficult, however, to overestimate how central a role CPR theory initiators assigned to the framework. Here is how Ostrom, Gardner and Walker underline it (*ibid*). "It is because we find complementarity in diverse theories and models that we need the IAD framework [...]. This framework provides a set of paradigmatic questions to ask, a metalanguage in which to ask them, and a spectrum of variable types for analyzing any microsetting. In order to use tools, one has to have a language about tools – their uses, strengths, and limits. To talk about any particular theoretical language, one needs a metalanguage." All theoretical and methodological resources used in CPR research fit within the structuring provided by the IAD framework. We would add that the framing of questions is important not only for discussing theory, but for practice too, as it both capacitates and limits our practical approach of management problems.

If we examine the summary of the framework as provided by Figure 8, two conclusions emerge that are essential in terms of our preoccupation of linking together economic tools with their contexts of use. The first is that the IAD framework is completely compatible with the founding assumptions and concepts of economic theory. It stages action situations in terms of agents, of their choices, their preferences, the information they have, how they can process it and how and with what results they interact with one another. The second is that, as the framework is set in very generic, "metalanguage" terms, it can also embrace all those approaches in social sciences that also rely on putting agents, their choices and actions, what they want, the information they have and how they process it, as the main concepts that frame their analyses. Many an organisation sociologist, a political scientist, an international relations scholar, an experimental psycho-sociologist, etc., would share that framework without a problem. Other social scientists would not, if they object to putting the rationales of individual interests and strategies at the centre of understanding social, political or managerial situations (chapter 8 will provide an in-depth discussion of such a perspective). For now, let us underline that, through the IAD framework, CPR theory links economic analysis and analysis of social and political interactions in a very explicit way, and at the deepest theoretical level. We shall see later in this section how the specific way in which the CPR effects this link creates both the high potential of, and the limitations to CPR as an approach to ETB use issues. Before that, a look at the theoretical and methodological resources mobilised by CPR scholarship will confirm and amplify this conclusion.

7.1.3. *Two tiers of theory, to be studied through mixed methodologies*

CPR theory combines two theoretical tiers. At the core is game theory reasoning: the study of situations where purely rational actors whose interests partly diverge are dependent on one another. There is a limited supply of fish; I want to catch more fish; you want to catch more too; I'll try to catch as many as I can, as fast as I can; but if we all do this, as our individual interest dictates, we will deplete the fish stock and will all lose in the end. In abstract terms, game theory is used in CPR to lay down a rigorous analysis of the structure of the dilemmas between individual interest and joint interest – the dilemmas we have to solve when we are trying to manage ecological resources held in common. This theoretical core is shared between game theory and economics, as suggests the very book that founded game theory: John Von Neumann and Oskar Morgenstern's *Theory of Games and Economic Behaviour* (1953).

The second tier prolongs the problem of collective action as outlined by Mancur Olson in his 1965 book on *The Logic of Collective Action: Public Goods and the Theory of Groups*. The core of the problem is the fact that even if members of a group would each and all have an interest in cooperating to solve a joint problem, this by no means implies that they will indeed cooperate. In our case, the fact that we may all have an interest in managing an ecosystem sustainably does not imply that we will, because we may be caught in the fundamental dilemmas of self- versus joint-interest described by game theory. However, such cooperation does exist, to a certain extent, in the management of those natural resources held in common that are actually sustainably managed – a point made very eloquently by a paper that has emblematic value in the CPR sphere and that underlies "the benefits of the commons" (Berkes *et al.,* 1989). To explain such cooperation it is necessary to add another tier of analysis to the first, game theory, tier. This is also required to guide the design of systems of cooperation to overcome the potential mismanagement of resources held in common – for instance, of ecosystem services that we may lose if each user just follows her individual interest. In other terms, we need to understand the forms of coordination that we can superimpose on game theory situation structures to "extricate ourselves" from the dilemmas of resource management. In the words of Ostrom and her colleagues (Ostrom *et al.,* 1994) "having reached the limits where modern game theory with fully rational players provides consistent theoretical guidance, we apply a theory of bounded rationality to explain the degree of cooperation reached among individuals who are given a chance to devise their own rules". A wide range of theories can be and are mobilised by CPR scholars to develop this second tier of CPR theory. There are a number of theories about how people sharing situations where they have diverging interests can do better than the outcomes predicted by non-cooperative game theory by communicating to agree on additional coordination rules and institutions. Such theories can come from a wide variety of academic backgrounds, from organisational sociology to anthropology, from law to institutional economics. This is no place for a review, but we need to underline two aspects.

1. However varied the theories involved, they remain a comparatively limited set because they can only be those that concentrate on "getting the institutions right", *i.e.* on "changing rules so as to improve outcomes" (*ibid*). This focus on rules and institutions is pivotal for CPR theory; it is a consequence of its two-tier construction. Since on the game-theory tier of the theory outcomes are determined by rules, interests and strategies, all the complex issues of the social, political, cultural, tier have impact only inasmuch as they impact rules, and thus, outcomes. As we shall discuss in more detail shortly, this very clear connection between highly complex and heterogeneous social contexts and resource management outcomes through rules confers the theory both its forcefulness and its limitations and connects it deeply with the economist's perspective.

2. The second (social, cultural, political) tier of CPR is extremely open and sensitive to the concrete context of action situations in all their complexity, their minute detail and their huge diversity. Anything that can impact the setting, implementation and outcomes of rules is relevant here. Depending on field situations, a huge array of factors can meet this condition. Furthermore it is not possible to hierarchise such factors *a priori*. As anyone with practical experience knows (or as any reader of field-researched biodiversity management and policy case studies soon finds out) what could in general seem to be details can, in a specific situation, make a management intervention succeed or fail. This reality is crucial for the analysis of the use of ETBs in actual situations and the fact that CPR theory squarely addresses it is a major aspect of its potential relevance. From a research perspective, it leads CPR research to adjoin to the very abstract approaches of the first tier (game theory, computer modelling) very qualitative and field-work-oriented approaches.

As we round up our presentation of CPR theory by a look at its methodological repertoire, this points to a remarkable feature of CPR scholarship: the deliberate combination of very different methods. CPR researchers apply various combinations of theoretical modelling, computer modelling, laboratory experiments with participants, computer simulations and role plays with participants in the field, and in-depth field monographs of CPR management cases. This is not an eclectic opportunism, but the quest of deliberate and careful combinations of methods (See for instance Ostrom *et al.,* 1994; Janssen *et al.,* 2011). Holding together that continuum of extremely different methods expresses into workable methodology the tension that, as we have seen, reigns between major tenets of CPR theory: (a) fundamental (*e.g.* game theory) structures of interests and rules are the central factor in explaining (or designing) resource management; (b) small details can make a huge difference in how rule systems play out in real situations; (c) thus it is necessary to study the multitude of heterogeneous details that affect coordination of interests through rules. Overall, to hold together the two ends of this very extended continuum, one has to deploy a high level of qualitative sensitivity to highly multi-faceted field situations while not conceding on the abstract readability of interest-rules structures: this is the fundamental reason why CPR researchers use such complex combinations of methods to bridge game theory situations and field situations of resource management.

7.2. The CPR perspective can help ETB users change rules to improve biodiversity outcomes

As we just discussed, in a CPR perspective the essence of action for better management of resources is "changing rules to improve outcomes". In the light of CPR theory, economic tools make a difference inasmuch as they are able to change outcomes for the better by changing rules, *i.e.* by replacing old rules, or by adding new rules.

7.2.1. *ETBs as propositions for changing rules*

A quick review of ETBs shows that they can certainly be analysed in this perspective.

Ecosystem services valuation can certainly be seen as a set of additional rules to the book of valuation methods. For instance if decision-making procedures on new infrastructures demand that cost-benefit analysis (CBA) prove profitability of projects, the additional requirement that the economic consequences of impacts on biodiversity should be included in such CBA is a modification of rules that may provide leverage for better consideration of biodiversity outcomes.

Payments for ecosystem services, as discussed in chapter 2, add one more kind of financial transfer in ecosystem management situations. This allows for one more kind of deal, on top of the many deals that already exist in such situations. The definition of PES by Muradian *et al.* (2010, p. 1205) as "a transfer of resources between social actors, which aims to create incentives to align individual and-or collective land use decisions with the social interest in the management of natural resources", fits perfectly with the idea of ETBs as rules added to resource management institutions and reiterates CPR's call for "getting the institutions right". The question: "Are PES effective for biodiversity conservation?" then translates into: Do such payment rules bring better biodiversity outcomes in real world situations? And just as in any case of CPR management analysis, it is safe to claim that the answer to that question depends at the same time on how a PES modifies the incentive structure (the economist's perspective) and on how it influences outcomes in practice, taking into account the details of the management situation in which it is implemented (the contribution of social scientists). An important addendum is that the quality and usefulness of the answer also depends on effective bridging of these two aspects.

As for the compensation and biodiversity banking family of ETBs, it fits most obviously the bill of CPR management as added rules [24]. By adding rules that set firm caps and require compensation, by setting further rules that allow exchanging compensations on a market, these systems have a potential to generate better biodiversity outcomes for a given level of development benefits. The caveat here is that, as we saw in chapter 4, for such systems to deliver tangible benefits they have to be part of institutional settings that are very strong in terms of legal obligations and of their implementation, of social and political commitment and pressure, etc. Of all ETBs, compensation and biodiversity-banking schemes may be those that most obviously consist in institutions squarely built on the purpose of halting depletion of a shared good, as ecosystems, habitats or species benefit from the institution of a no net loss principle.

So far, that is, on the conceptual point of seeing economic tools as changes or additions to rules, CPR theory and economics are on the same page: outcomes are determined by agents pursuing their interest in the framework of established rules. If we want to improve outcomes, and if we cannot change directly what people want, we must change the rules. This leads us to the next challenge: what does it take to effectively change rules in real life management and policy situations? As strong as economic theory is in considering in detail and theoretical depth what kind of changes in rules would improve outcomes, it is not equipped to address the next step: how do you effectively implement rule changes in real contexts? CPR theory has invested much effort in clarifying and addressing this question and there lies the main contribution it has to make as we strive to move towards actual, relevant and effective use of ETBs in the field. Its treatment of what it takes to change the rules combines the two tiers we introduced in our presentation of the theory: respectively game-theory and field-situation sensitive. To discuss it, we shall proceed in two steps, the first with a stress on the first tier, the second combining both and stressing the second tier.

7.2.2. *How do you change the rules of a game while you are playing?*

Here is how Ostrom, Gardner and Walker (1994, p. 300) summarise their conclusions from both theoretical and empirical enquiries.

"(1) Rules shape action situations, including situations that can be represented as games. (2) Rules shape action situations by affecting the incentives and choices available to individual actors, to which rational actors respond by adopting certain strategies and

[24] The reader may have noticed that we have skipped land and right acquisition tools. They are discussed at the end of this chapter.

behaviours, which affect outcomes. (3) Changing rules can therefore change action situations in ways that motivate individuals to adopt different strategies and behaviour, potentially yielding different outcomes. (4) Rule changes can be developed and deliberately chosen by the actors in the action situation, as well as imposed from outside. (5) Actors in an action situation change the rules shaping that situation by taking action at multiple levels [see below]. (6) Actions frequently occur not only at multiples levels within a particular action arena but also in linked action arenas."

We have already covered points 1 to 4 – which expand on changing rules as the crux of action for sustainability – in the previous section. We would just like to underline one important point at this stage by drawing on Ostrom's analysis that effecting a change of rule does not differ from creating a new, original institution. "Both origins and changes in institutions can be analysed using the same theory when both are viewed as alterations of at least one status quo rule." (Ostrom, 1991, p. 140). This echoes an issue that has been recurring in the first four chapters as we reviewed ETBs: can just one tool, just one new rule, change biodiversity management from a failure to a success? "Introduce monetary values and nature will be taken into account in decision-making"; "pay for provision of ecosystem services and people will continue (or start) to produce them"; etc. The difficulty, the challenge is the same whether one sets out to create an institution or to change one by adding new tools – thus new rules – to it.

As we move beyond the naive view that we could create institutions where there are none – that ETBs could operate by themselves as if in a social, political and institutional vacuum – the question focuses more appropriately on the conditions under which adding a new rule can generate sufficient change to solve a given biodiversity problem. Since "getting the institutions right" is the motto of CPR management, for ETBs, the question becomes: to what extent can a given ETB, in a given resource management situation, set the institutions right? In practice, when dealing with a case of ETB use, what is needed is a comprehensive diagnostic of the resource management institutions as they are, and as they could become, modified by the adjunction of an ETB.

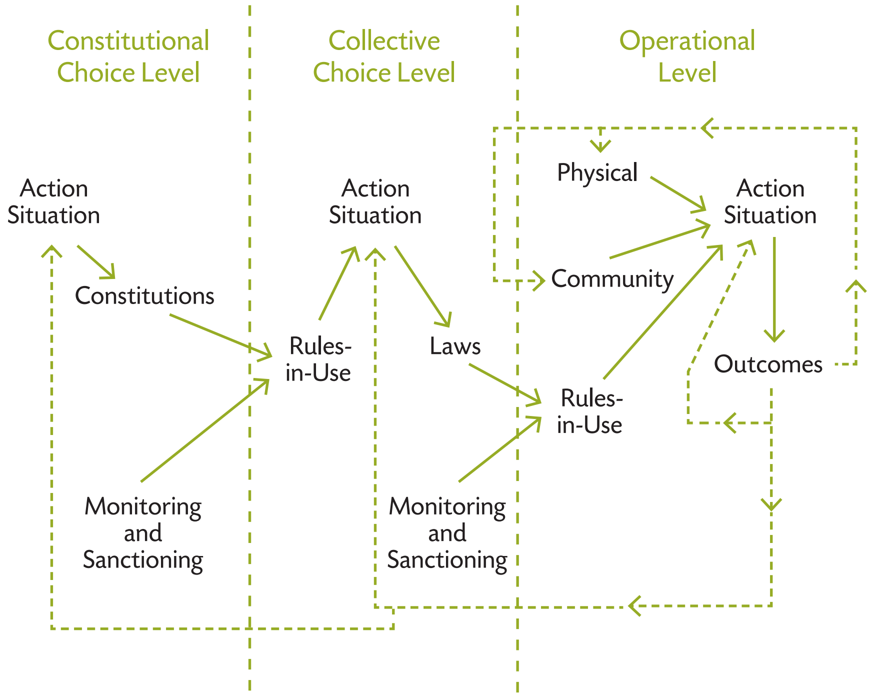
On the bright side, it is clear that there is potential in changes of rules – such as adding ETBs to existing ecosystem management systems – to improve many biodiversity outcomes. It is true that in some cases the changes may have to be very deep, changing substantially the nature of the games that are played with the resource. But in numerous other cases, limited changes in the rules, or additions to the rules, can make a great difference to biodiversity.

On the darker side, however, is the fact that changing just one rule (and having that change effectively implemented on the ground) may be extremely difficult. If the rule does change management, it will almost always negatively affect the interests of some users. If it brings substantial changes, it is likely that these impacts will be strongly felt and that affected users will resist the change. Even if the impact is not great, but the affected users hold a lot of power, resistance to change can be very high. Reading cases on biodiversity conservation, or on ecosystem services degradation, the story comes up over and over again of attempts to change rules in the face of mismanagement and of a backlash of resistance to such change. Knowing what kind of rules could allow us to operate sustainably is one thing. The question of how we can act to effectively change rules is quite another.

As reflected by items (5) and (6) in the long quotation that introduced this section, the way CPR theory approaches the difficulty of changing rules is linked with the preference of CPR scholars for the resources users "*extricating themselves* [25]" from their situations of actual or potential misuse rather than waiting until they might be helped out by someone else. This implies a fundamental paradox: the users operate within the current set of rules which plays a decisive part in their choice of strategy regarding the resource. How can they change the rules while playing within the rules? To extricate itself from the paradox, CPR theory differentiates the rules users play by, and the rules about changing the rules. More precisely, they separate three types of rules: operational rules that govern day-to-day choice (*e.g.* no boat on this fishery can fish more than ten days in a season); collective-choice rules that bear on how and by whom the operational rules can be changed (*e.g.* on the fishery, the owners of authorised boats will meet annually to revise maximum number of fishing days according to previous year catch); constitutional rules that bear on how and by whom collective-choice rules can be changed (*e.g.* if the rules for authorising boats on this fishery have to be changed, this will be done by a committee of elected authorised boat owners, selected experts and government observers) (cf. Figure 9).

[25] Our emphasis.

Figure 9 *The three levels of rules / Linking levels of analysis*



Source: Ostrom et al., *1994.*

In other words, resource users play several games in parallel: they play within the rules and they make moves about modifying the rules. As Figure 9 shows, the changes in rules are played out by the users in view of how the operational rules affect the outcome. The central assumption here is that users who compete in appropriating the resource can cooperate to change the rules of that competition in order to obtain better collective outcomes.

It must be underlined that in a CPR perspective, this is not just an assumption for analysis, it is also the formula for action towards sustainability. A CPR-inspired blueprint for improving the use of ETBs for biodiversity will have to rest on expecting that biodiversity stakeholders will make a cooperative effort to introduce new operational rules and tools bearing on their activities, so as to improve jointly desirable biodiversity outcomes. How can ETB experts help them on that route?

7.2.3. *Two levers for change: increased communication and information on behaviour-outcomes relations*

Resource users may be in need of help because the route to improving rules is fraught with difficulties. Maybe on a conceptual level the distinction between levels of rules does reduce the contradiction between users being caught between competitive resource extraction and cooperative rule-setting. But it does not by itself resolve the tension: we would still need explanations on how cooperation to change the rules is produced. The main answer of CPR theory to that question can be summarised in one word: communication (Ostrom *et al.*, 1994, p. 149; Janssen *et al.*, 2011). Janssen, Bousquet and Ostrom, insist that communication *per se* – the amount of communication – is more important than the content of communication. This reflects three important aspects of CPR scholarship. (1) The first belongs to the first tier of the theory, as it stems from game theory. In game theory, the limits put on communication are a major trait of non-cooperative game behaviour. Thus changing communication patterns are of the essence if one is to overcome the traps of non-cooperative dilemmas (Shelling, 1960). (2) As one moves to the second tier, with its sensitivity to complex contexts of action, it becomes apparent that the modalities and contents of useful communication are extremely variable depending on context. As a result, on a general, theoretical level one can mostly retain the general principle of increasing communication (and leave the question of how to field applications). (3) As CPR theory bridges the two tiers, however, there appears a largely implicit, but very powerful assumption that there is a thematic core to communication content. It consists in the hypothesis that discussing the relationship between rules and outcomes is instrumental for fruitful change, since the whole stake of communication is to change the rules so as to improve outcomes.

Based on this reasoning, two sorts of contributions can be expected from CPR experts and practitioners. The first is to facilitate communication, like a facilitator, or a mediator. The second is to make as explicit as possible the structure of the action situation and the links between rules, behaviours and outcomes, and then to leave the users to find the appropriate ways to improve the management system. The obvious synergy between the two is that if there is much communication between the resource users, they are more likely to search actively and jointly, and eventually to find such improvements. So quite often experts strive to combine both in methodologies and to link facilitation of communication between users and modelling of behaviour-outcomes relations. Again, discussing the feedback loops between outcomes and action situations (see figure 9) is the main source of insight for the users to communicate and decide about possible changes in rules.

It is striking to see, as we examine how the CPR approach frames change towards sustainability, the great extent to which a similar framing and conception of the experts' role already dominate the ETB literature. This is particularly evident for ecosystem services valuation (EVS). Many writings in that field are underpinned by the (mostly implicit and unquestioned) theory of action that (1) valuation brings awareness of the negative outcomes of neglecting biodiversity and (2) confronted with the evidence, the stakeholders will in some (non-described) way cooperate to change behaviour. As for economic tools (PES, acquisition, biodiversity banking), there is also the widespread persuasion that (1) if research and pilot experiments make it clear enough that outcomes can be improved through the use of the tools and (2) if such information is widely and actively made available, then stakeholders will find ways of adopting them and so change rules, behaviours and outcomes.

The insight and experience gained in the CPR field can be useful to improve ETB use here. It demonstrates that producing information or tools that may make better outcomes reachable with appropriate changes in rules is not sufficient to bring about change. It must be complemented by intense communication between the stakeholders about the rules-outcomes connections. Furthermore, such game-changing communication is impeded by the interests associated with the existing set of rules and by extraneous obstacles (social, cultural, political, etc.), so that to overcome such barriers, an active involvement in provoking and sustaining communication is required.

Building evidence on rules-outcomes connections and facilitating communication are very different activities and require completely separate expertise. ETB users could benefit much from the vast body of expertise that exists (both in the CPR field and elsewhere) in terms of methods and practice in facilitating communication between biodiversity stakeholders. The two activities may be done by the same or by separate experts. The important point is that if an economic tool for biodiversity is to be a "game-changer" – a phrase that may be trite but that reflects quite accurately the bottom-line of CPR theory – they both have to be carried out effectively and to be linked up with one another. This conclusion concurs with the movement of thought that promotes participation in ESV and in the implementation of economic tools in the field. We should point out however, that the kind of participation expected to make a difference, from a CPR perspective, has to be communication between those whose activities have a significant impact, and such that they can eventually change the rules. These terms of reference mark out the way that leads to the use of ETBs by biodiversity stakeholders themselves.

They are very demanding terms of reference indeed, but realistic ones. The CPR literature makes it very clear that good management of the commons is a demanding task, and that it often fails. A favourable combination of factors is necessary. Going back to success stories like the Vittel and the Bushtender programs – and many cases that are repeatedly used as positive examples of ETB use – it appears that they involve a context where many factors favouring successful implementation (such as the wealth, power, motivation and strong local connections of the company producing Vittel water) are present to an exceptional level, and aligned to create a very good governance situation to start with. And even in such favourable contexts as in the success cases that are most often discussed, it obviously took a lot of effort and ingenuity to make the PES or the biodiversity banking scheme a success. Conversely, it would be unrealistic to expect that in a biodiversity management situation where many factors concur against their potential effectiveness, tools from the ETB panoply could be expected to make a "game-changing" difference.

This calls for case-specific diagnostics. In a CPR perspective, the central aim of such diagnostics is (1) to determine whether the action situation corresponding to a biodiversity issue is such that it is realistic for the stakeholders to be able to create and implement new institutional rules that will deliver improved outcomes and (2) to examine what role the economic tools one considers using might be relevant as part of such new rules. The CPR literature can provide much guidance in that direction. Good places to start can be the basics of CPR theory and the list of "design principles" we quoted at the beginning of this chapter. From these can be derived sets of relevant questions like the following, to investigate a situation where users damage biodiversity: Is it the case that all or most users would be better off if they could extricate themselves from excessive free access? Is the situation such that stakeholders and users can close access to outsiders effectively enough for rules restricting their own access to allow good management of the resource? How favourable to open discussion and negotiation amongst the stakeholders are the social and political conditions in the case? And what would it take to improve such communication? What are the external (economic, administrative, political) pressures on the users and stakeholders? How realistic is it for the latter to expect to be able to check such pressures sufficiently in order to maintain an effective system of rules for managing the resource?

Benefits of sound diagnostics of ETB use situations on CPR lines would be (1) to apprehend clearly – both overall and in detail – what issues are standing in the way of successful creation by the stakeholders themselves of sustainable biodiversity management institutions, so as to be able (2) to assess to what extent one can expect an ETB scheme resting on stakeholders mobilisation to make a decisive difference, and (3) to draw on a detailed situation diagnostic for use in such a scheme's design.

Obviously, rule-making by resource users themselves is only one scenario of ETB use. Referring to the previous chapter, it is one based on "coordination" of the actors, and one could just as well envisage other scenarios based on "government" intervention to make users change their ways. This will then require other forms of field diagnostics, but these will have to look for other foundations than those offered by CPR theory, which is inherently about self-ruling by resource users. But before we turn to discussing the limits of the CPR theory to conclude this chapter, and to other perspectives in the next chapters, we would like to insist on the importance of field diagnostics because the CPR field has been particularly active and ambitious in that area.

7.2.4. "Consolidate principles, multiply case studies": can the ETB field emulate the dual ambition of CPR scholarship?

To this point, we have insisted mostly on the theoretical contribution of the CPR field. Another contribution that can be very useful for the analysis of ETB use situations is the field's vast experience and remarkable organisation regarding case studies. Since CPR theory insists on the decisive importance of local circumstances, its promoters took the logical step of setting up a system to encourage and collect case studies of common-pool resources management. Over the years, literally thousands of cases of managing fisheries, rangeland, underground water resources, forests, etc. have been collected and discussed. They form the basis for the pragmatic design principles we presented earlier, and they allow detailed discussion of what makes various arrangements work or not in a given case. Let us insist again though, with CPR authors, that beyond the generality of design principles, each field case has its unique combination of factors that can make a given system of rule (like a given ETB) succeed or fail.

This concurs with what has been a recurring conclusion in our first four chapters reviewing ETB use: in every field case, the overall multi-faceted management situation has to be diagnosed in all its complexity, and the contribution of the ETB (to be) used in the case has to be established in full view of that complexity. If we think of the Vittel case in the PES chapter, or of biodiversity banking in the Bushtender program in Australia, the ETBs used do not deliver the ecological outcomes by themselves, but as components in a much more extensive and complex management (or public policy) strategy. We are well aware that repeating such statements can be frustrating for the reader as they may be felt as just stating "it's more complicated than that, we can't tell you much up front, it will be necessary to establish a new analysis of the case from the ground up"!

Well, the point is, the issues involved in ETB use are actually that complex and case-dependant, and the whole ETB field will need to deal with it. CPR scholarship sets a very useful precedent with the way it has approached this issue, *i.e.* by organising an ongoing confrontation between a very clear and structured framework and theory on the one hand and on the other hand an ambitious, open-ended system of case studies that fosters well-founded case-by-case analysis and recommendations for practical management. This would be a very useful orientation for the ETB field as use is now becoming the crucial issue for their ability to make a difference for biodiversity and is also highly dependent on local situations. The way to go is to combine (1) an effort to lay down very explicit frameworks and theoretical resources for the analysis of use issues and (2) a systematic investigation, publication and discussing of ETB use in specific biodiversity management situations. Encouraging and collecting hundreds, even thousands of documented case studies [26] would be an extremely useful way for the ETB field to embrace a serious study of use issues and to sustain the duality of powerful general principles and infinitely varied local circumstances that is inherent in using a management or policy tools like ETBs. As for ETB users confronted with their particular biodiversity problem, they will each, regarding their own case, have to confront general frameworks and design principles with the unique circumstances of their problem.

In the confrontation of general principles and case-based circumstances of ETB use, CPR provides only one perspective. After pointing to its usefulness in the previous section, let us now turn to its limitations.

7.3. Probing the limitations of CPR theory as an approach to analyse and treat biodiversity issues

There are, indeed, serious limitations to be considered in the use of a CPR approach for the analysis and treatment of biodiversity issues. Some stem from the fact that CPR theory is centred on resource management and thus misses important aspects of biodiversity. Others are inherent to the theory's focus on users' interests and the resulting difficulty in grasping externalities. Still others are linked to the fact that CPR management resolutely relies on a symmetric, "coordination" approach and thus restricts itself to only part of the wider spectrum of collective action we reviewed in chapter 6.

[26] This may sound overambitious, but the CPR network has done it, and considering the number of researchers, of master and doctoral theses focusing on biodiversity field issues globally, it is in fact mostly a matter of organization and of the ETB field's priorities.

Before we move on to examine successively these three sets of issues, we would like to point out that their relevance for ETB use extends much beyond the scope of CPR alone. The intellectual foundations of ETBs have such proximity to the CPR approach that a discussion of the limitations of the latter carries over useful insight on the limits of the reasoning underpinning ETBs and especially – and here lies our interest – on how it affects their use for actual biodiversity management problems. Indeed, much of the discourse on ETBs is based on a two-tier reasoning similar to that of CPR theory. The first tier is provided by environmental economics, which calculates the links between rules, interests and outcomes – and how hypothetical rule changes could change these. The second tier consists in an exploration of possible changes of rules that would lay the foundation for the implementation of economic tools. We shall see that, as they share deep-seated similarities, the CPR approach and the environmental economics of biodiversity can share a useful discussion of the former's limitations.

7.3.1. *Does sound resource management amount to good biodiversity management?*

CPR is essentially about renewable resources. The bulk of the literature and of the practice it relates addresses a limited set of resource-management fields: fisheries, forests, water, hunting, wild vegetable products collection and the management of extensive production systems like range management (which are combining aspects of ecosystem management and of agricultural production). There is evidently a strong overlap between this set of resources and biodiversity. Mismanagement of the resource – deforestation, over-fishing, over-extraction of water, over-hunting, bad range management – causes negative impacts on biodiversity. But can sound management of the resource be equated with good biodiversity management?

On the one hand, good management of resources is often a prerequisite for biodiversity conservation and management – for instance when community forestry both prevents massive logging through industrial forestry and allows recovery of forest habitats. And many major biodiversity problems are caused by unsustainable resource management (low-nature-value [27] agriculture, overfishing, deforestation, etc.) So by all means CPR management in its own terms can be an important resource in building solutions to many biodiversity issues. But is it a sufficient solution? The

[27] We propose this phrase to mirror the concept of 'high-nature-value agriculture'. It reflects the fact that there are indeed farming systems and practices that damage biodiversity.

need for clarification is quite high here, since many authors implicitly or explicitly present sound CPR management – and more generally sound rural development – as the main avenue to treat biodiversity problems.

Good resource management is not sufficient for good biodiversity management because many aspects of biodiversity are of little or no relevance for resource users. For instance, rare, inconspicuous orchid species in European peatlands are hardly of concern to those who can use and maintain such peatlands as resources – rangelands, watershed heads for water management, or landscapes to be conserved for tourism. Hundreds of yet-undiscovered insect species in an unlogged tropical forest do not concern its various users. To treat such biodiversity problems, appropriate land use is a necessary condition – and again, CPR management can contribute to that – but is by no means sufficient. Range management can conserve the landscape and water qualities of the peatlands while wiping off the rare orchid species. Sustainable logging of the tropical forest may be much better than clearing it off for commercial agriculture, but it may damage insect biodiversity. A great many biodiversity issues are dependent on, but additional to, appropriate resource management.

Three scenarios are then possible in terms of how the management of such biodiversity issues that are not directly connected to resources can be linked with sound resource management.

1. A certain resource management system may provide some biodiversity benefits contingently, without the managers' and users' intention to do so. Much of the biodiversity of rural areas is the result of resource management systems that let it live, or that provided favourable conditions for it. The issue here is that when such management systems change, for whatever reason, this may threaten biodiversity. Indeed, resource management does change with time. A resource management regime that provided good biodiversity outcomes as a by-product can change and come to produce quite poor biodiversity outcomes under pressure, for example if users become more and more numerous with the same individual needs, if the technology they use evolves, if the markets they rely on to turn resource into income become less favourable, etc. Seen from the point of view of resource management, of the users, and of those actors who support them for development and poverty alleviation reasons, such changes may very well not be objectionable at all. So there is a necessity for some biodiversity issues to be treated as specific issues, distinct from, additional to, and sometimes conflicting with sustainable resource management. This is where the other two scenarios come in.

2. One first way to take into account biodiversity issues that are left aside by resource management is to get to treat them also as resources on the basis of the benefits that they provide to the users. This is the logic of ecosystem services: the idea of integrating them along with other commodities and other services to make them part of an extended picture of resource management. In a CPR perspective, however, where resource users have to negotiate the management rules between themselves, this requires that individual users do feel the benefits (or potential loss) of good biodiversity management. The very driving force of change in CPR (and more widely in resource management) is that many users fear a tangible loss for themselves if some biodiversity item is lost.

In practice, this may be the case for biodiversity items that generate tangible benefits or that have remarkable cultural salience. It is bound to leave aside important parts of biodiversity. And if this is true when such elements generate no tangible benefits for the many, it is even truer if their conservation requires some sort of effort or sacrifice in resource exploitation (for instance if conserving tropical forest insects requires set-aside of some forest tracts).

3. As a result, many biodiversity items can simply not be taken charge of without specific measures being put in place aside from and in addition to resource management. This is important because as one discusses the usefulness and the possibility of integrating biodiversity issues in resource management, a heated debate arises on whether it is legitimate to keep alive a capacity to act in favour of biodiversity independently of resource management and resource managers. That debate is made more sensitive by the fact that actors specialised in interventions for biodiversity often have to play the dual role of overcoming the shortfalls of resource managers in terms of managing ecosystem services, plus taking charge of biodiversity items that fall outside a resource and service perspective. Without venturing further into that debate, let us simply state here that as we move into sounder biodiversity management across the board, there will have to be a part of biodiversity that is taken in charge through resource management, and a part that remains the responsibility of networks specialised in biodiversity that is neglected by, or comes in conflict with, resource management.

A careful consideration of these three scenarios has important consequences for the use of ETBs, because three kinds of use have to be considered. (1) In the first, economic tools are used to help put in place and sustain a sustainable resource management system, *e.g.* common-pool resource management, because such a system, on top of its benefits for resource users, has biodiversity benefits. (2) In the second, economic tools are used to practically integrate biodiversity items and ecosystem services into a resource management system that becomes extended in this way so as to cover a wider array of resources. An example could be PES schemes where some users (of water from a forested watershed) pay other users (of forest products) to limit their extraction of wood to sustainable levels, as part of an overall set of rules for common management of the forested watershed. (3) In a third configuration, economic tools are used as elements of policies and management that target biodiversity, independently of, or in addition to, the management of resources. This is the case for example when payments are distributed for additional requirements that are not of direct concerns to users, but are intended to ensure specific biodiversity outcomes – for instance when agri-environmental payments compensate farmers for employing suboptimal resource management techniques, such as late grass harvest, that deliver superior outcomes in terms of corncrake reproduction.

Taking this into account, diagnostic for better use of ETBs should really combine the answers to two questions: What would the conditions be for sustainable management of the ecosystem-based resource? What specific additional conditions have to be met to deliver specific biodiversity outcomes?

7.3.2. *How far can the CPR framework go in tackling environmental externalities?*

The latter question is not really a resource management question, but rather an environmental one: it implies that someone other than the resource users cares about ecological impacts generated by resource use, and this pushes us to the limits of the CPR perspective. At the very onset of her writing on CPR, Ostrom underlines that "there are limits on the types of CPRs studied here: (1) renewable rather than non-renewable resources, (2) situations where substantial scarcity exists, rather than abundance, and (3) situations in which the users can substantially harm one another, but not situations in which participants can produce major external harm for others. Thus, all asymmetrical pollution problems are excluded, as in any situation in which a group can form a cartel and control a sufficient part of the market to affect market price." (Ostrom, 1991, 26)

As one considers using the CPR framework for environmental problems, the fact that it considers many externalities to be out of its scope is indeed a serious issue! Certainly, "producing major external harm for others" is the core of environmental issues and of the conceptualisation of environmental problems, *e.g.* through the concept of externalities in economics. Biodiversity is no exception here, if one considers the massive damages to biodiversity caused by global-scale industries (in forestry, farming, mining, fisheries, etc.).

The way this issue is usually dealt with is by recommending that one create wider communities of stakeholders so that instead of "others", those who are affected by negative environmental outcomes become part of such communities, and that those who produce the impacts change their attitude and start caring about them. The French concept "*patrimonialisation*" is interesting in that it combines the meaning of heritage (something to be handed over to future generations) and of asset (a capital good that can be used to produce revenue or services). "*Patrimonialisation*" is indeed the action and process of making-a-good-a-common-good and applies to cultural heritage, landscape, and any sort of biodiversity object. For instance, "*une espèce patrimoniale*" could translate as "a heritage-asset species".

The whole thrust of much of the literature in that field is to promote transforming situations where the ones, by their impacts, harm the others, into situations where they become members of the same, extended community, working to coordinate themselves for better environmental management institutions through intense communication (for instance with the support of mediation, of collaborative approaches, of participatory conservation processes, etc.). Such an approach is tantamount to a social, political and institutional internalisation of externalities. If successful, it may indeed create a context in which economic tools for internalisation of externalities can play their part. In this direction may lie an interesting lead for ways to better embed ETB schemes in their context. A great example is the Vittel case, in which the first diagnostic commissioned by Vittel was sociological, bearing on perception by farmers of their profession, of other actors, and of their relations, the next steps – producing the science of how to get the right outcomes and designing the economic tools – having come only at a later stage as additional resources to help the newly created, latent, community of actors organise concretely a new management of the land and water in the Vittel watershed.

But however promising the attempts to transform situations of externalities into communities of co-management, they cannot always be expected to overcome the challenges of serious situations of externality and asymmetry. The same factors that facilitate the transition to commons management– such as the small geographical scale of the resource management situation at hand, the particulars of a given ecological and technical situation and their fit with given management solutions, tangible material interest for users, symmetry of interests in users, or cultural and political conditions (see table 6) – point clearly to those factors that make such a transition very difficult: large-scale systems, complexity and poor understanding making it difficult to link rules, behaviours and outcomes, lack of material interest in the issue for actors, vast differences between the actors, their power, their activities and worldviews, etc. These are very tangible realities which determine the actual presence – or absence – of margins of manoeuvre for actors to create a successful commons regime.

In many situations, such margins of manoeuvre may be very limited, or non-existant. As Brendan Fisher *et al.* (2010) show on the basis of two cases of PES schemes for watershed management in Tanzania, often one is so far from fulfilling the necessary conditions for commons management schemes that one can come to question the feasibility of such schemes. In their case studies, these authors find that the geographic and political scale may be too large for successful implementation, that groups of users are very heterogeneous, that they do not have a very direct relation with the resource nor the capacity to produce expected outcomes, that the cultural and administrative context make payment rules hard to accept and implement, that there is a low level of institutional structuring of resource management in the area and that there are intense external pressures playing against sustainable use of the ecosystem and the resource.

We concur with the authors as they conclude that "care must be taken to make sure that the lessons we learn while heading down the PES path were not already learned in other contexts, with other literatures, and other buzzwords. Here we think that the opportunities for PES systems to learn from CPR management are great." (*ibid*)

7.3.3. *Limits stemming from the underlying coordination paradigm of collective action*

The third and last point we would like to discuss about the limits of CPR approaches stems from their view of collective action. Referring to the five paradigms of action for change we reviewed in chapter six, CPR theory sees resource management problems as coordination problems. Solutions do not primarily come from government, nor from intensified dialogue between government and stakeholders (except for government to tolerate or support problem resolution by the stakeholders themselves), and very little is said in the theory about minorities who act for change; as for revolution, it is not a strong presence on CPR theory's horizon. Not that these other paradigms are excluded completely, but they are not the operating basis of CPR theory. The operating basis is definitely coordination between users or stakeholders. More precisely, coordination is conceived as collaboration in setting up institutions, a collaboration which is seen as the best way for users to extricate themselves from the dilemmas of resource management. CPR theory does concur with numerous approaches to sustainable development of the environment that are based on cooperation to establish shared management and shared rules. It is indeed hard to refuse the concept that it is worthwhile to seriously try to go as far as possible in building collaborative solutions to biodiversity issues. There are, however, important limitations to what can be accomplished by direct efforts to encourage collaborative processes. Let us examine two of them more closely.

The first is the treatment of issues that are a concern to only a minority of stakeholders, especially if they are not resource users. Since the main organising force in a CPR perspective is collaborative rule-making by the users, the fact that a majority of users have shared interests and perspectives is decisive both for the possibility and the content of the rules they will design. Conversely, CPR approaches are not relevant for problems that are of concern only to a minority of users, or can be treated as resources only in some far-fetched manner. For the many biodiversity items that fit this description, one must look to other approaches.

The second limitation results from the essentially collaborative theory of change that underpins CPR approaches. Consider again Figure 9, which depicts how, from a CPR perspective, the understanding of the outcomes from a given set of rules are used jointly by resource users to change the rules so as to ensure better joint outcomes. This is the crux of the theory's approach to change. Two conditions are required for it to operate. (1) The fulcrum of change is the fact that individual users will benefit from higher joint outcomes. Otherwise, let's follow the reasoning of a user who would not benefit. Remember: he is reasoning based on his own outcomes. It would make perfect sense for him to use any means at his disposal to obtain changes in operational rules. For this, it would make just as much sense to interfere with the establishment of collective-choice rules and, higher up the line, of constitutional rules. In other words, it is perfectly logical to expect non-cooperative-game behaviour not only within the game, but in rule-making too. (2) The lever of change is the capacity of users to suspend for a moment their immediate competition within the game, and reconsider the game itself. Now if some users draw large individual benefits from a game that is badly designed from the point of view of joint outcomes, and that the long term viability of such benefits is not a concern for them, what is a rational behaviour for them? A reasonable answer could be: to oppose clarification and reflection on the game by any available means; to resist proposals for changes in the rules.

Anyone involved in biodiversity issues can evoke from their experience numerous examples that this is not just abstract reasoning. Sector-based interests and lobbies do exert great pressure to slow down clarification of environmental problems, to stop environmental rules from being put into place, or to obtain that those already in place be relaxed. The strategies and the sheer power of some actors or sectors to unilaterally appropriate certain resources whatever the consequences for biodiversity and for other users, and their capacity to neutralise collective action efforts to the contrary are an incontrovertible reality across much of the field of biodiversity conservation.

Most CPR scholars are fully aware of non-cooperative behaviour – it is precisely the reason why they invest so heavily in collaboration. However, they do not address the issue in a frontal way, but rather indirectly. Rather than focusing on facing the power of destructive activities to create the conditions of collaborative rule-making for management, they focus on those situations where building as strong as possible cooperation may overcome the perils of non-cooperative strategies. The "design principles" that serve as a leitmotiv in the CPR literature (local scale, users affected in a similar way, little uncertainty on consequences of actions, no damaging interference from States with local ruling, etc.) are a list of conditions that favour collaborative action gaining strength and thus indirectly limiting the possibilities of non-cooperative behaviour. They can just as well be taken as a list of the conditions that would prevent solutions based on the collaborative setting of sustainable-use rules: large scales, users with vastly different interests, large uncertainty on the consequences of action, active involvement of States against local ruling, etc. There is a limit to the amount of unilateral strategy and power pressure collaborative efforts can take. CPR theory has great relevance within that limit; it has quite a lot to say about operating on that limit, where collaboration is possible but difficult. The domain beyond that limit – in our view, a vast domain in terms of biodiversity issues – is out of CPR theory's and practice's purview.

Limits of CPR approaches in addressing environmental externalities and limits of cooperative approaches are really two aspects of the same problem: beyond a certain level of asymmetric power pressure, the cooperative creation by stakeholders of commons institutions is no longer a viable proposition. Other strategies, based on other diagnostics that address more directly power issues are then necessary. We shall discuss them more particularly in chapter 10.

7.3.4. Elements of common-pool resources management in situations of private or public ownership

For now, setting aside those cases that are either clearly within, or outside the domain of user-initiated rules, it is worth examining more the grey area of situations where there is an element of negotiated rules, without common property of the land or resources. This is particularly interesting to understand ETBs that rely on buying land, or on buying land-based rights. At first sight, they seem rather disconnected from a CPR pattern, since they rest squarely on private or on public property, not on rights jointly owned by users. And this is indeed the case when, for instance, the State buys land and then manages it itself, in a top-down administrative way. But is this necessarily the case?

In chapter 3, we examined the example of the French "Conservatoire du Littoral", through which the State buys coastal land to prevent its development, but then has to devolve management to local public or private operators, typically for multiple use management. Such management relies heavily on multi-party negotiations and management plans. Often, it has just as much in common with managing land that is partly held in common as it does with top-down management of State land. Our general point is that this category of tools, just as many regulatory conservation area tools, essentially operates by blocking some of the most biodiversity-damaging management options, in this way creating space for other stakeholders to negotiate management rules that provide a diversity of uses and good conservation outcomes.

The fact that land publicly bought or legally protected creates space for commons management systems and processes is often overlooked for three reasons. First, there is a widespread confusion between the most explicit logic of legal tools and the real processes of their implementation. The fact that in theory one could exercise complete discretion in one's decisions does not mean that in practice this is always possible, or advisable. Protected areas regulations do not enforce themselves automatically. The owners of land bought for conservation do not manage it in a social and political vacuum. Where the principle would seem to be unilateral decision, the actual practice is multiple and intense negotiations. Second, the more localist approaches to resource management, like CPR theory, were born in an opposition to State policies and to private property as a solution to resource management issues. As a result, the synergies between State policy, private property and commons management are caught in a sort of theoretical no-man's-land and are largely overlooked. Third, Finally part of the literature on resource management, because it lays closer to rural development interest than to conservation ones, also relays the discontent of some local resource users. From the fact that these have not obtained all they wanted in negotiations *e.g.* because they have had to make some concessions for biodiversity's benefit, it concludes that there was no genuine negotiation, no agreement, and no commons management. This is too narrow a basis for a sound diagnostic of the outcomes of a decision-making process and it tends to obscure the large presence of negotiated resource use rule-making in settings that have also a strong presence of private property, of State property, or of protected areas regulations.

On the basis of years of investigating biodiversity cases, we do believe that such mixed situations are very numerous indeed. For a relevant understanding of many ETBs, it would be very useful to cast aside simplistic dichotomies between private property, State property or rule, and common goods, and to invest seriously in the investigation of the complex synergies between State, ownership and commons management that play such a central role in biodiversity management. This is particularly relevant for ETBs. As we reviewed them in chapters 1 to 4, again and again we have observed that multiple patterns intertwining economic, social, administrative dynamics are at the crux of the real-life use of ETBs.

Conclusion

CPR theory provides important resources for better diagnostics of ETB use situations. (1) It provides theoretical clarity and coherence, with deep-reaching theoretical roots. (2) Its fundamental capacity to link together economic and social, political and institutional reasoning is an important asset. (3) The great and still growing body of work it produces on forests, fisheries, forestry, range management and other resource issues that are key for biodiversity gives it a wide sphere of relevance. (4) Its attempt to push as far as possible efforts for joint rule-making may be a no-regret option in many situations.

For the development of ETBs, there are particularly two kinds of applications where CPR theory could be used more than it currently is. The first is the diagnostic of given ETB use field situations. Where ETBs amount to adding new rules involving money to status quo institutions, overall and detailed understanding of the whole system of rules, and of the cooperation it relies on, is of the essence. The second kind of use would be a critical reflection on the theory of action that is widely shared in the ETB literature, according to which a clear understanding of the links between economics and biodiversity, plus intense communication, would constitute in itself a powerful way to bring about better cooperation in forging better rules, at all scales, for biodiversity conservation. We would need to use CPR theory, and in line with the spirit of CPR scholarship, to be extremely explicit about under what conditions this theory of action is relevant in the real world. We would need to back this up with empirical investigation of how given biodiversity issues are actually managed at all scales. This may help us take the measure of how far the equation "explicit costs and benefits + intense communication = cooperation on new rules = the way forward to sustainability of ecosystem services" can take us toward effective biodiversity conservation.

There are, however, three blind spots in the CPR perspective. (1) It is heavily centred on a calculus of users' interests. Those non-interest-based social and cultural elements that are taken into account are seen as secondary variables affecting the conditions of the interest-based game that it considers to be the core of resource management. This may limit the extent to which CPR approaches can investigate non-interest-based dimensions of resource management, which are such an important aspect of biodiversity issues. (2) In all the CPR literature, conditions of relative closure (clear boundaries, clear community of users, etc.) are a major theme and condition for successful commons management. This is inherent in the game-theory and communication-based foundations of CPR theory. But it limits its capacity to explore open, global networks of influence that are such important drivers of contemporary transformations in the economy and in biodiversity at all scales. (3) The focus of CPR approaches on cooperation in rule-making limits its capacity to account for, and deal with, the powerful forces that deploy strategies to prevent or counter collaboration for biodiversity conservation.

Our discussion of paradigms of action for change in the previous chapter has shown that coordination can hardly be the only perspective to consider when examining our capacity to act for change. As write Ostrom, Gardner and Walker (1994, p. 49-50), "Given the organizing character of a framework it is more difficult for scholars to work across different frameworks. [...] We do not address the question of alternative frameworks within this volume as we have not yet encountered problems where the IAD framework is not a useful tool for addressing policy problems. However, we do not presume that the IAD framework is the only framework available to social scientists interested in understanding questions of social order." In the next three chapters, we will examine three other possible frameworks, very different from IAD and its CPR declension for resource management. Each addresses one of the blind spots of CPR we just listed: justification theory is pointedly not centred on actors' interests; actor-network theory is interested in ever-expanding networks of actors and in initiative for innovation; strategic environmental management analysis focuses on the power dimension of the struggle for biodiversity.

8. Clarifiying the ethical challenges that besiege biodiversity economics: justification theory can help

Our whole approach on economic instruments for biodiversity is driven by a pragmatic concern: how can we make the tools work to the benefit of biodiversity, not just in abstract situations of economic theory, but in the multidimensional messiness of real-life management and policy situations. As we engage the ETB literature in search of approaches and debates centring on use of the tools, again and again we find challenges and debates about the normative issues raised by ETBs. (1) The most head-on confrontation is from authors who oppose on the one hand the sphere of economics and the sphere of ethics on the other hand. In that perspective, promoting the use of ETBs can be seen as intrinsically immoral. (2) Less radical and more frequent, but just as challenging, are views that express concerns about a balance between market-based and other foundations of our policies and management systems. A central theme here is commodification, *i.e.* the idea that grasping a biodiversity item in terms of economic valuation or tools transforms that item into a commercial good. For such critics, endowing a biodiversity item with a market value strips from it other, deeper values: aesthetic, spiritual, affective, traditional, etc. Also, as it provides a stronger grasp on biodiversity items for operators with financial resources, they criticise the fact that it disempowers, at least in relative terms, those who have little money or no official property rights: the poor, the indigenous, local people who use ecosystem services or have an affective link with ecosystems. In the eyes of its critics, commodification thus has two faces: a more abstract one that reframes our thinking in terms of tradable goods, a more practical one that actually hands over more power to market operators to the detriment of other people. John O'Neill (2007) argues forcefully that both are synergic: by thinking about a biodiversity problem in a market framework, we lose our grasp on other dimensions and we drift towards practical market appropriation. (3) Beyond these tensions between the market values promoted by ETBs and other important social values, other critics insist that what we need in order to deal with the biodiversity crisis, and more generally with the ecological challenges of our time, is a set of new values clearly centring on ecological systems and on the links between humans and nature. From that perspective, ETBs are also seen as problematic, because they promote the same market values that underlie our current social and economic order, which critics see as part of the problem, not of the solution. For them, even if short-term pragmatic gains can be obtained from the use of ETBs, such benefits would be undermined and eventually outweighed by the reinforcement of a damaging value system.

Overall, the sheer pressure of ethical challenges directed at ETBs is such that it would be quite problematic to try and hold a serious discussion about the pragmatic issues of ETB use without engaging in the normative debate that they set off. Furthermore, far from being confined to academia and its literature, perplexity and controversy about the normative dimension of ETBs also pervades the practical scenes of their use. In our interviews with field practitioners, the issue is raised again and again. As one of them says, we have to ask "What are the ethics surrounding these things?". And when questioned about the kind of issues raised by the practical use of ETBs in their organisations (consultancies, NGOs, etc.), they often mention first the internal debates raised by the very principle of using ETBs. And again the issue is raised when they are asked about how they identify situations for which the use of ETBs would be relevant or not. An NGO person asks: "How do we sell valuation as a concept to countries objecting to it? And should we do so?". And an economist involved in development aid policy underlines that "we operate in a time where the issue is to define a new system of values". He adds that many economists show up with technical solutions to a problem which, today, is not technical; that markets reflect a social contract, and that the contract that has remained relatively stable since World War II now needs renegotiating; that in this time and context, one of the most useful contributions of ETBs may be to embark ourselves and to embark field actors with us in an exploration of workable value systems in the situations we are in.

To sum up, such is the diversity and intensity of ethical challenges addressed to ETBs that a serious treatment of their ethical dimension is a necessary part of working on their use in management and policy. To that end, we will here use justification theory. It was expounded in 1991 by Boltanski and Thévenot in their master book *De la justification – les économies de la grandeur,* translated into English in 2006 . Soon after its first publication, researchers realised the potential of the theory for the analysis of environmental issues and since then it has acquired a large influence in France. Indeed the theory addresses one aspect of environmental issues that is very salient but seldom treated in depth (especially by technical and economic approaches): the plurality and conflict of incommensurable values that are confronted when dealing with environmental conflict and decision-making. Moreover, it does so in a way which is consistent with commonly held notions of the different normative logics that clash in such situations – industrial against traditional, commercial against aesthetic, etc. – but that has much clarifying power due to its rigorous theoretical construction and to its firm grounding in empirical evidence. We have adopted it here as the theory or choice to address the ethical dimension of ETB use for the following three reasons.

First, it very explicitly removes self-interest and strategic reasoning from the centre of attention, and quite squarely concentrates on the ethical dimension. Luc Boltanski and Laurent Thévenot, the initiators of the theory, insist that as long as one does not restrict oneself to noticing mostly strategic behaviour, it appears that in many cases people strive to set difficult situations in order to look for principle-based justification to assess options. They insist that it does not do justice to these efforts of ethical justification to treat them as just opportunistic accessories to strategic positioning. Quite the opposite, they should be taken as a fundamental dimension of human behaviour. They are especially important in the organised treatment of public problems, as occurs in most situations of ETB use.

Second, the main focus of justification theory is on how people actually work at coordinating themselves, in real-life situations, through moral justification of behaviour and decisions. This makes it particularly fitting for our particular concern in this book. Indeed what we are most interested in here as we approach ethics, is to find intellectual tools that will help us better analyse the exchanges of critiques and justifications as they unfold in practical situations related to ETB use.

A third reason for our adopting justification theory as our choice approach to grasping the ethics of ETB use is simply the fact that in the context of environmental management and policy issues the theory is already quite powerful and has a large potential for further work. Admittedly, it is currently almost unknown in the English-language literature on environmental issues – just as obscure, one might say, as Common-pool resources theory which we discussed in the previous chapter is omnipresent [28]. But we think the effort to be made in discovering it is fully justified by its combination of immediate relevance, high systematicity and theoretical depth. We shall see that it covers and articulates together, in ways that are relevant for practice, the various ethical challenges to ETB use we have met in the literature and in our interviews.

[28] This may be explained in part by the comparatively recent date of publication of the English-language version (2006), combined with the fact that taking up the theory requires a significant investment of time and reflection.

And as we will show, there is a large potential for new developments of the theory to expand its treatment of environmental issues.

In this chapter, we shall start by a presentation of justification theory. We will then see that it sheds a new light on the ethical connections of economic valuation and tools, as it allows moving away from frontal opposition between economics on the one side, and ethics on the other. The next section will examine how ETBs embody a variety of "compromises", *i.e.* combinations of incommensurable orders of worth, and how understanding such compromises serves the debate on ETB use. Finally we shall try to move beyond the orders of worth initially identified by Boltanski and Thévenot. We will examine whether ecology implies a specific order of values, or is adequately relayed by the currently dominating orders of value. We will also question the new values underpinning the project-based organisation of action which is so prevalent in the current implementation of ETBs.

8.1. Justification theory: how to legitimise actions in presence of multiple orders of worth?

Justification theory belongs to the wider perspective that is now labelled "pragmatic" and has dominated French social science since the 1980s (Dosse, 1995). A central tenet of that perspective is for the social scientist to break away from his critical, "overhanging" position, *i.e.* from any position that amounts for the researcher to assuming that he holds the key to understanding the actors' situations and behaviours, whereas the actors themselves operate under misleading interpretations of their own behaviour. This is for instance the case when a social scientist presents actors as hiding interest-based behaviour from themselves and from others under the cloth of general principles of morality – an interpretation that tends to suggest that the cloth of purported morality is shoddy, and that the real motives at play are just cynical interests. For "pragmatic" social scientists, people know first-hand what they are doing, so that we should abandon such "overhanging" perspectives and devote our attention to following how people give meaning to the situations they face, and how they deal with them on their own terms. The role of the social scientist is then to help describe and clarify the multi-faceted efforts of the actors and the ways in which they constantly transform situations, rather than to superimpose from the outside meanings that are alien to the situation.

In the case of Boltanski and Thévenot's justification theory, the focus of attention bears on how people criticise one another and justify themselves to one another. For the social scientist to observe this from a critical standpoint is problematic: to critically analyse the actors' behaviour, the social scientist has to discard their normative standpoint and look at the situation, as it were, from the outside. This hypothetical "outside", however, is not less normatively loaded than the actors' own standpoints. The "overhanging" position of the social scientist creates a situation where he would be the only protagonist in the debate who would not have to justify his own positions in the face of critical challenge (Boltanski, 1990). This might still be acceptable if the researcher were the only one to have the ability to critically analyse behaviour and discourse by stepping outside the normative framing set by the actors. However, observation of how actors deal with issues ranging from small to large shows that they spend much time and skill doing precisely this. They step outside of each others' framings and expend a lot of skill and effort in criticising each other through an analysis of their respective discourse and behaviour. Rather than positing himself as a critical analyst, and viewing actors as (at best) unaware or (at worst) devious strategists, the social scientist should rather move from critical sociology to a sociology of the actors' critical behaviour and capacities. This is precisely what Boltanski and Thévenot set out to do in the research that led to justification theory.

Their approach has been both empirical and theoretical. On the empirical level, they have observed disputes, for instance, arising from conflicts in the workplace (Boltanski, 1990). In such disputes, actors try to justify their own behaviour and to criticise the behaviour of others. To succeed, both justification and criticism must be considered credible by many or most of the protagonists and witnesses of the situation. For this, the actor has to show that what he claims to be right in his particular case would also be considered right in all cases with the same circumstances. In other words, critical or justificatory claims have to be backed up by principles that have a high degree of generality. Studying efforts to criticise and justify leads to searching for widely-shared normative principles that actors are using in confrontations associated with practical situations. An observation that stands out is then (1) that a variety of incommensurable principles are used by actors as they criticise-justify but (2) that only a limited number of such principles are regularly used. The latter observation is consistent with the fact that justifying a claim requires that the principle used be shared by many, and that there is likely to be only a limited number of widely shared principles, as opposed to the unlimited range of idiosyncratic individual normative references. This dual observation leads to the question: would there be a set of widely shared, incommensurable scales of values that are used by actors in disputes of all sorts? This question lays the foundation of justification theory.

To proceed on the theoretical level, Boltanski and Thévenot started from the idea that, if such widely-used scales of value indeed exist, the actors who use them have to have learned them. So there should be ample evidence of these values and principles being discussed, shared and taught. Boltanski and Thévenot set out to find and analyse texts that would expound such scales of values and be used as resources by actors. In *De la justification,* they analyse two such sets of texts. The first is a series of "know-how" books used by people for advice about behaviour in the workplace – for instance a union-published guide on the rights of workers and the associated procedures, or a sales manual. The second is a series of widely-known and respected reference books in political philosophy – for instance Adam Smith's *The Wealth of Nations* or Jean-Jacques Rousseau's *Du contrat social* that each puts forward a coherent model of how society can and should order itself. These two levels of reference texts illustrate the move from the most particular cases to the most general principles that actors make when they criticise-justify and that the social scientist follows and clarifies as he observes them doing so. Someone experiencing a workplace problem argues to justify his own behaviour; in doing so, he looks for arguments that may have leverage; he seeks advice on such arguments through discussions; these discussions are fuelled by teaching, by readings, etc.; in turn such teachings and readings appear as applied and practical forms of systems of values that have been written in abstract, theoretical and very general form by others.

Through this approach, Boltanski and Thévenot have identified six systems of values, six orders of worth. Each of them has a widely-based legitimacy in our society and they are all massively used in critical-justificatory debates about issues of all sorts on all scales.

1. The *civic* order is based on common interest accessed through political and legal procedure.
2. The *market* order is based on the active search for trade-offs that are profitable for the protagonists.
3. The *industrial* order is based on efficiency.
4. The *domestic* order is based on tradition, on a hierarchy of personal links of dependence.
5. The order of *fame* is based on the degree of attention from others.
6. The order of *inspiration* is based on the capacity to access altered, inspired states of mind, as in artistic creation for instance.

If this just reflected the observation that often justifications based on tradition clash with market rationales, or that the quest for efficiency puts pressure to shortcut civic procedures, there would be little more here than common sense. Indeed, being a sociology of how people handle their critical-justificatory arguments, it is to be expected that justification theory has common ground with common sense. But it goes much further than just identifying basic diverging discourses of everyday life and management. It shows that they express orders of worth that have been collectively constructed over the centuries, that are widely shared and deeply engrained in contemporary culture. These orders are not just orders of ideas and argument. They also underline the concrete organisation of co-existence and ethics-based shared action. In a way, they are political orders of worth, and Boltanski and Thévenot name them polities (*cités*) to underline that their logics are at the crux of the way we deal with public disputes and strive to construct the common good. For them, at the core of contemporary societies lies the capacity to manage public issues despite – or thanks to – the co-existence of several, firmly established and incommensurable orders of worth. We all share several orders of worth, although they contradict each other, and it is precisely the tense interplay between them that organises the space within which we articulate and justify our individual and collective decisions. Justification theory breaks away from the temptation to refer all issues to just one, overhanging, order of worth. But it equally stays away from the idea that if there is not one such central order of worth, then what we have is a fragmented landscape of heterogeneous values we should renounce dealing with in a systematic way. From a justification theory perspective, there are a plurality of orders of worth, but to be able to withhold the pressures of public dispute, they need the strength and legitimacy that result from a solid collective construction and dissemination and so remain quite limited in number.

For Boltanski and Thévenot, the six orders of worth they have identified are built on the same architectural principles – in their terms, they share a common "grammar" of what constitutes a viable system of worth for public justification of a position. The crux of that architecture is dealing with the tension that exists between two contradicting demands. On the one hand, a principle of fairness and equality demands that everyone be treated on the same basis. On the other hand, people, situations, things, have to be ordered – *i.e.* hierarchised – in terms of worth, and thus put on an unequal footing. The key to a legitimate order of worth is that inequalities between people due to such ordering have to be justified by the fact that this specific hierarchisation does contribute to the common good, a contribution that Boltanki and Thévenot call a shared overarching principle (*principe supérieur commun*). For instance, the efficiency motivated constraints of a production organisation and the resulting inequalities in practice may (or may not) be justified by the resulting efficiency in production contributing to the common good of participants in the production process. Another element that joins the two contradicting demands is the investment – the sacrifice – that is expected from those who endeavour to elevate themselves in a given order of worth. To move from "small" to "great" in one order, one has to focus one's efforts in a way that leads to sacrificing some other aspects of life. This is no place to elaborate further on the details of the architecture of justification theory's orders of worth. For more in-depth use of the theory, we can only refer the reader to Boltanski and Thévenot's exposition, and turn here to introducing yet another component of the theory that will be of great importance in our analysis of ETB use.

The strong presence of these orders of worth does not mean that all people's actions are led, organised, even less determined by them. Justification theory does not see the confrontation of various orders of worth as rigid and mechanical. The polities are not confronted in a vacuum. They are used by people to resolve issues in real life situations that are inevitably composite. They also do not provide a way to classify people or organisations. For justification or criticism, the same person can refer to one or another polity according to the situation at hand. The capacity of the same people to appreciate several different sets of values that may apply in different situations is an integral part of their pragmatic competence. This is definitely not a theory that holds a stereotypical view of orders of worth associated with rigid, stereotyped logics of action. For instance in their presentation of the theory, Boltansky and Thévenot show very clearly how a company combines aspects of each of the polities they describe: the industrial and market ones of course, but also the domestic (allegiance, loyalty and pecking orders established over time) and the visionary (the leader with a vision, the creative "madman"), the fame (reputation) and the civic (stakeholders and the public acceptability of the firms activity). In day to day situations, all these dimensions have to be tended to in resolving issues within the firm. More generally, the interactions through which actors seek justifiable resolution of issues are caught in a tension between on the one hand the necessity to justify choices based on widely accepted principles (justice) and on the other hand the necessity for solutions to be appropriate for the practical situation at hand (*justesse*).There is no divorce of values and practice here, but a rigorous analysis of the use of values in and for practice.

Also, many, even most human actions, as Thévenot (2006) analyses, do not fall into the category of actions that have to seek public justification. But sometimes they do: they are, in the theory's terms, put to test. And it is in such test-situations (*épreuves*) that protagonists have to resort to arguments of worth to justify their own actions or criticise others'. Typical examples of such test-situations would be for instance environmental pressure groups challenging a government infrastructure project, or media attracting attention on resource users that are creating environmental damage, or again forestry interests attacking a national park project on the grounds that it destroys development and employment opportunities. It is easy to see that situations where ETBs are, or could be used, are precisely such test-situations when values – and not just interests – are pitted against one another or are laboriously searching for reconciliation. Not just that ETBs would be, as it were, innocent bystanders caught in the midst of environmental value struggles, like workmen at a construction site powerlessly observing a row between competing architects and waiting for the conclusion to go about their own, essentially technical job. ETBs are themselves essentially about assigning value, so they are fully involved in environmental value-testing situations.

In such situations, arguments have to hold under critical pressure and so they have to be grounded in robust orders of worth. It is the parallel operation, in our societies, of several deeply rooted orders of worth, that are at the same time robust and partly incompatible, that structures the space of moral debates associated with the critique and justification of publicly made claims and decisions. Boltanski and Thévenot's six orders of worth and their "grammar" (the concepts that describe the foundation of robust value claims) can help us map that terrain and get a clearer grasp of some of the main challenges of value debates associated with ETB use. As we examine them in turn, we will introduce additional elements from justification theory that illuminate specific issues.

8.2. Deciphering the crossfire of value-based critiques

The most immediate application of justification theory is its ability to clarify exchanges of critical arguments premised on different orders of worth. Boltanski and Thévenot's book devotes a long section to a "table of critiques". They systematically review, on the basis of their six management textbooks, how each order of worth founds a specific critical reading of each other order of worth. They show us how the critique of the market order seen from the civic order is now well established and "can express itself in the laconic form of catchwords, as in the reference to *capitalism*, or in the opposition between owners (*the egoism of owners*) and *workers*. [...] It is also evident in the pointing out of *deviations* that threaten collective entities when particular interests prevail over the quest for the common good" (p.319). Or the other way around, how the domestic order viewed from the market order appears as an unjustifiable system of inertia, of traditions, prejudice, routines, that block the development of fruitful opportunities; in that light, interpersonal relations, social ties and local loyalties appear as unfair obstacles to individual freedom, initiative and exchange. Or again, seen from the industrial order the instability of the market order is a source of disorder, as it threatens rational planning (which requires stability in time) and may favour the superfluous (luxury goods for instance) over the useful. Beyond such examples, it is the capacity of justification theory's model to map such critical exchanges in a systematic manner – six orders of worth criticising each of the other five, *i.e.* thirty well-definable critical viewpoints that can be activated in controversies – that allows it to encompass our common-sense grasp of the crossfire of critical perspectives, while elevating it to a higher level of explication and precision.

Indeed, as we read about, or listen on the controversies about ETBs through the lens of the six orders of worth model, we clearly see one after another the possible combinations being activated. A good example is provided by critiques addressed to biodiversity banking [29]. Take the case of a development project that destroys a wetland, for instance, and the developers buy compensation that will take place at some other location. (1) This will in no way compensate the loss felt by nearby residents, who will suffer a loss in their ties to the land and in day-to-day amenities that they took for granted as they had always enjoyed them. (2) Neither will it compensate for the aesthetic value that the land may have had, nor for spiritual connections. (3) Furthermore the residents will not have their say in the compensation scheme, which is decided in the context of a commercial activity and of bureaucratic planning that compensation may help escape citizens' critique and grasp. (4) Finally, there is no way that compensation can guarantee an absolute replacement in ecological terms. The loss of one place's biodiversity is going to be replaced by something that will not only be somewhere else, but that will be somewhat different. Depending on the type of ecosystem at stake, on the type of compensation scheme, and on the care and expertise invested in running it, the difference may be more or less pronounced, but there will always be one and thus, in absolute terms, a loss. The reader will have easily identified critiques directed from (1) the domestic, (2) the inspiration and (3) the civic orders respectively. The fourth item is less clearly covered by Boltanski and Thévenot's original six orders of worth – we will turn back to this matter further on, as we discuss the question of a specific ecological order of worth.

[29] For instance as expressed by John O'Neill during a seminar organised by the research project that led to this book.

The more we train ourselves to understand the prevailing orders of worth and the more clearly we see each one's internal logics, the less such critical controversies will appear to us as volleys of contingent attacks, and the more we will understand that they express the structural combinations implied by the fact that contemporary societies operate on the basis of relatively stable but partly incompatible scales of value. As one acquires a clearer reading of them, one can become more skilled at participating in and at moderating the value-based confrontations about biodiversity issues. But a more immediate conclusion is that there is no way to think away the confrontation of irreducible coexisting value systems.

What then is to be made of John O'Neill's stringent critique of biodiversity banking? Not that biodiversity banking schemes are never appropriate. Absolutely no development project would pass the test that it would disturb no existing habits and links, that it would effect no aesthetic change or that it would cause no change whatsoever in ecological systems and anyboy's life. In our view, the point that is really made by that critique is that reciprocally, there is no basis for giving biodiversity banking *carte blanche*, as if the sort of compensation it offers should make the affected projects acceptable *a priori,* regardless of the specific circumstances of each case and the associated confronted values.

More generally for the ETB user who wants to develop and use a given tool, this means that however clever the tool's general design, he cannot claim that the tool's use ought to be generally accepted. The possible use of the tool will have to undergo every time the test of being confronted with the specific value claims of a given problem, of a given place and of the affected people. ETB users will have to engage again and again in the exchange of justification and critique that gives parties to a decision the opportunity to test its normative soundness. Accepting such confrontation as legitimate (not as a tiring and useless aside of serious business, nor as a personal offence), preparing oneself for it, are an integral part of readiness for the use of ETBs in real-life situations.

8.3. Re-thinking the usual opposition between economic tools and ethics

This questions in return to what extent one is ready to recognise that economic reasoning and tools also have some moral standing. Indeed, if in the confrontation of values attached to each case of ETB use the ETB user would be the only participant without any ethical standing, then the confrontation of values could only be sterile and often violent. This is a significant concern, as in the ETB literature, economic reasoning is often opposed to ethical reasoning as if economics and ethics were disconnected. This opposition is implicitly present, for instance, when O'Neill and Spash (2000) discuss the problem of articulating economic valuation and tools on the one hand and ethical concerns on the other as an important issue both in the design and in the implementation of ETBs. They describe two main methods for this articulation: the capture of ethical values in economic valuation (for instance through willingness to pay for non-use of biodiversity), and what they call the "moral expert" approach, for instance through ethics committees, in which economics and ethics are dealt with separately through distinct procedures. They criticise both, in particular for failing to be democratic enough, and recommend a more deliberative approach, in which all parties involved in an environmental decision would be able to express their values and hold a debate both on the practical issues at hand and the values to be referred to in deciding about the case. In various guises, this type of solution – complementing (or replacing) economic tools with participatory procedures – holds an important place in ETB debates and practice today.

The explication of the content and construction of the value claims at play can certainly help facilitating and participating in such deliberative procedures. And particularly, so would clear indications on how economic values and ethical values (or diverging ethical values) relate to one another.

Justification theory makes a framing move of major importance in this direction by breaking away from any crude opposition between economics on the one hand and ethics on the other. As writes Boltanski (1990; see also Thévenot, 1989), "people participating in a market are *moral beings*[30], in that they are capable of setting aside their particulars so as to agree with one another on external goods, the list and definition of which are universal". In other words, rather than sticking to his own perception of an object, the person on a market accepts a public test of the worth of the object as agreed upon by others through a public procedure. This acceptation of submitting one's own valuations of things to public testing in view of shared orders of worth is precisely, for justification theory, the foundation of ethics. There are also less abstract grounds on which to establish the moral character of economic values. "The market world is peopled with individuals striving to satisfy desires. They are in turn clients, competitors, buyers or sellers, having business relations with one another" (Boltanski and Thevenot, 1991, p.247). The moral principal at play consists in moving,

[30] Author's emphases.

exchanging goods and services to satisfy desires: not just one's own, but those of others too. The transaction which is at the root of any market relation is a reciprocity of desire satisfaction that involves private benefit, but also a form of common good – Adam Smith's "invisible hand".

Critics of market-based valuation and tools might argue that the market order of worth is just a facade for private interests. They can indeed find convincing examples of this. The point is that any order of worth can be used as a strategic device to provide an ethical facade to the defence of particular interest. Technical arguments can be diverted to favour solutions that provide private benefits to some – which is a central theme of the treatment of social issues by sociology of science and of the critique of technologies. Presenting some activities or interests as traditional may barely cloak private or sector-based interests – another classic of biodiversity and resource management issues. In the field of biodiversity, the use of "domestic" arguments (based on tradition, livelihoods) by agricultural or forestry interests to cloak corporate or bureaucratic strategies is a major factor of confusion (for detailed examples see for example Rowell (1996); Benhammou and Mermet, 2003). There are not on the one hand market-based arguments that would always cover some private interest, and on the other ethical arguments based on tradition, on civic procedure, etc. that would automatically stand for the common interest. Each order of worth can be used both as rhetoric to back up for an interest based claim, and as a scale of value to promote one of several forms of common interest.

Again, to recognise the specific dynamics of confronting publicly various scales of values applying to biodiversity, one has to renounce the attitude that would see all behaviour, all arguments as just strategic and self-serving (what Boltanski and Thévenot, following Paul Ricoeur, call "suspicion" (*soupçon*)). This may sound difficult in some contexts where social science is dominated by such attitudes of critical suspicion. But it is important to realise that this renouncement of purely strategic readings of situations is required also by deliberation theory. Recommending to base the treatment of public problems on more deliberation (public participation, open governance procedures, etc.) requires that one credit participants with a certain capacity for sincere value-based discussion, and not only for strategic designs. This is not to say that strategic behaviour does not occur, is not important, but only that it is not the only dimension of public controversy and deliberation. Simply, just as no one (including the public and local affected parties) is devoid of strategic design, no one should be considered without any ethical standing and capacity to participate in a public debate – a debate that inevitably articulates private interests and the common good together.

So economists are entitled to having ethical concerns too; what do they consist in? In our long-standing participation in controversies about ETBs, we have been struck by the frequent expression of moral indignation by economists when an economic transaction that would generate wealth is blocked on the basis of other arguments (tradition, regulation, public pressure, etc.) that plead for the status quo. Such reactions are sometimes dismissed by critiques as an expression of arrogance, or of a will to make the logics of economics systematically prevail over any other consideration. But could it not be interpreted also as the clash between orders of worth? Since these are incommensurable, if someone defends seriously one set of values he may well be felt by others who are attached to other, contradictory sets of values, as being obstinate and overbearing. Whether you call it development, free-market or poverty alleviation, trading things, services or rights to satisfy more human desires – the foundations of the market polity – may not be the alpha and omega of human existence but is nevertheless the basis of one very significant order of worth in the organisation and values of our society.

One of the contributions of justification theory to the analysis of ETB use is that it sets very explicitly the market as one system of ordering values, amongst other systems of ordering values. No one system is, nor should be, in a position to claim priority in general. Which one should prevail has to be discussed in each situation when people have to reach publicly justifiable agreement. In such situations normative points of view are often defended with passion against each other. Between the strong assertion of an order of worth in the context of a controversy, and the claim that it should prevail over others whatever the context, there can be a thin line. Distinguishing the two, separating the assertiveness of advocacy from the finality of a closed ideological worldview, is however essential. There is a world of difference between claiming that economic values should prevail and economic tools be used whatever the context, and advocating that economic values and tools should be taken seriously in debates about how we can manage biodiversity.

The report on EVS by the French Strategic analysis council, cited in chapter I.1 provides a good example of the former attitude, when the report recommends using EVS only for "ordinary biodiversity", because the members of the expert group authoring the report concluded that "exceptional biodiversity" is considered so for reasons (aesthetic, a feeling of heritage, a movement of public opinion, etc.) that belong to other orders of worth and that decisions about it should be referred to these other scales of valuation, rather than to ESV.

Another aspect of the line separating advocacy from imposition is the attitude of ETB advocates to the defence of other orders of worth. Paying lip service to the existence of other orders of worth but ignoring completely the people who advocate them in practice cannot count as sufficient pluralism. One has to engage in constructive dialogue, in a collective effort to combine different orders of worth and sets of tools for the treatment of the biodiversity issue at hand.

To sum up, justification theory puts forward an explicit and sophisticated model of how the market order of worth co-exists with other orders of worth in such deliberations, as one system of justice-seeking values amongst others, not more, not less.

8.4. ETBs as "compromise" tools

To this point in the discussion, however, we have assimilated ETBs with the market order of worth, as we presented the debates between the market with other orders of worth. Does that discussion apply directly to ETBs, however? Do the economic valuations and tools used for biodiversity belong entirely and simply to the market? Opinions differ on this point in the ETB literature. In their paper entitled "What's in a name", Emma Broughton and Pirard (2011) discuss the label "market-based instruments" that is often attached to ETBs and shows that their connections to markets are highly variable, and often very loose, whereas they are also closely linked with administrative or legal systems, as we showed about biodiversity banking in chapter four. Robert Constanza (2006) insist that valuing ecosystem services does not imply that they should be managed through market systems, whereas O'Neill (2007) maintains that the very principle of economic valuation carries with it a market logic that undermines essential political and ethical dimensions of managing environmental issues. Justification theory can help us sort out this debate, and thus make one more step towards clarifying debates on values in the use of ETBs.

For this, we need to venture a little deeper into the theory by examining with Boltanski and Thévenot the different types of interactions and deals that may occur when people defending different options confront each other in a decision-making situation.

The first question to examine in such circumstances is whether the protagonists share the same order of worth as reference, or whether they disagree (implicitly or explicitly) on the order of worth that should be used to judge the situation. Consider a company preparing to lay off part of its work force. As managers discuss who should go, they may diverge even if they use only criteria belonging to the industrial polity – *i.e.* if they reason based on the demands of planning for effectiveness and efficiency. For instance, should they keep experienced personnel, so as to minimise problems caused by the diminished workforce in the short run? Or should they keep younger higher potential new arrivals to speed up the renewal of the work force? But they may also diverge on the very order of worth that should be used. One of them may want to keep the younger part of the work force while others may consider, based on a "domestic" logic, that the firm has a loyalty obligation to those workers who have been there longest, are part of the local community and have fewer options for alternative jobs.

Boltanski and Thévenot insist that these two configurations are deeply different. In the first one, the issue becomes one of ensuring that an appropriate test is applied to the situation. In our example, what data and criteria are appropriate to decide between two concerns both pertaining to the quest for efficient production? The test must have quality and purity. Quality in that it must provide relevant and accurate ways of assessing options in view of the shared values – in our example, by providing a relevant comparison of options in terms of their capacity to minimise the loss of productive capacity. Purity in that it must not be contaminated by logics from other polities – for instance, if the principle has been chosen to implement the option that best minimises productive capacity loss, and if one manager insists on keeping a worker that has family links with other workers in the factory, this will be felt to taint the test and will create a difficult situation if decisions are made in a context where they have to be publicly justified.

In the second type of configurations, there is simply no common scale which the protagonists can use to reach a strongly principle-based agreement in judging the case. We have seen above how each order of worth is highly critical of the hierarchies promoted by each of the others. But as the exchange of criticism has to give way to practical decisions one will have to overcome the incommensurability of values to reach a decision. Two distinct configurations can then occur.

The first, Boltanski and Thévenot call a "compromise" (*compromis*). Linking together elements from one and from the other of the two polities that are confronted, the protagonists build an explicit, hybrid set of principles. Such compromises may not be as robust and universal as the polities. They can, however, be considered as a regional form of validity and be used repeatedly to solve similar issues. They will remain vulnerable to criticism that would demand that a purer hierarchy of values be used. But they will nevertheless be sustained to a degree by the acceptance they may have achieved and by their practical usefulness. Such compromises are an important aspect of the analysis of situations in the light of justification theory.

In other cases, however, the clash between diverging orders of worth is such that the protagonists do not succeed in constructing a normative compromise that would allow them to justify their choices in view of clearly articulated principles. If they still have to make a deal they will have to renounce justifying it in normative terms, based on principles of worth, and will assign it only to the necessities of the particular situation at hand. Justification theory calls such deals accommodation (*arrangement*).

To sum up, three main configurations have to be considered: (1) debates on the purity of tests within one order of worth, (2) discussions to elaborate compromises, *i.e.* hybrid systems of principles and (3) sheer accommodations without normative foundation. Typical of the first are debates amongst economists on the extent to which various methodologies and tools conform to the demands of economic theory. The third is certainly not inconsistent with some ETB use situations. After all, money is a very flexible medium and can be used in all sorts of deals that have no normative defence to present for themselves. But the second is much more interesting here because it helps us understand better some of the complex normative debates that underlie the design and use of ETBs. In the first four chapters of the book, we saw that the various types of ETBs were not used in the rarefied atmosphere of pure economic theory but in complex combinations with ecological expert advice, administrative rule-making, etc. As we examine them in a justification perspective we will find that they are best understood as relying on hybrid normative constructions – Boltanski and Thévenot's "compromises".

8.4.1. *ESV as an industrial-market compromise*

Let us start by examining the orders of worth that appear in the notion of ecosystem services. The most salient is the industrial polity. As a concept, ecosystem services sets the focus on the capacity of ecosystems to produce goods and services, on the efficiency of such production, which is approached through comparisons with other ways of producing similar services. Integrating services rendered by ecosystems into reasoning on the optimisation of goods and services production processes has been an important stream of ecology and environmental thought ever since the 1970s. Ecological planning is a clear expression of the industrial polity, as it integrates ecosystems in planning efforts which strive to make the most rational use of space and resources for the production of goods and services of all sorts. Adding economic valuation to the analysis of ecosystem services brings in the monetary dimension. But in ESV, money is not only a being of the market polity. It is also used as a device to compare the efficiency of alternative processes for the production of goods and services because it reflects closely many of the inputs that enter in the "industrial" efficiency equation: amount and availability of materials used, amount of work consumed, amount of equipment and infrastructure needed. Overall, the main order of worth underpinning ESV is that of an industrial-market compromise.

It is by far not alone in so doing in the environmental field, as illustrated for instance by the collective pollution-abatement funding systems put in place through the French watershed agencies, as analysed by Godard (see Narcy, 1994) using justification theory. Such framing is an explicit move to set the treatment of ecological issues in the same normative framework that underpins most of contemporary development policy: the effort to satisfy more needs through the combination of more efficient production designs (industrial) and intensified trade (market).

8.4.2. *Reactions in quest of purity*

The hybrid, compromise nature of ESV, however, is a constant source of challenge and irritation. In that respect, it is interesting to remember the movements of thought that, since the 1970s, have striven to replace monetary valuation of ecosystem services by direct, physical analysis and quantification. Using energy as the main unit for establishing equivalence between various processes, both man-made and natural, has been a favourite. In our view, these efforts have not been very successful, because they fail to integrate the variety of the components of service producing processes at play and of elements to be factored in when establishing the worth of an ecosystem and its services. Many aspects of biodiversity tend to be missed in such analysis. It may even be counterproductive. If we managed ecosystem based on maximising stocks and flows of any component – be it energy, be it CO_2 – we may be led to destroy, rather than to conserve and restore much of the biodiversity that has, in many ways, value for us. But these energy-based analyses illustrate well the normative strength of the industrial polity.

Repeated efforts to replace monetary valuation by other units of measure to assess the efficiency of ecosystem services illustrates very well justification theory's point about the unease felt in situations where tests of worth in one polity (here, industrial, the efficiency of ecosystems in producing goods and services) is mixed with criteria of worth from other polities (here, the market, through the adoption of money as a language). This affects the purity, and thus the legitimacy of the test. It is this purity that efforts to value ecosystem services in units that clearly belong to the industrial logic (such as energy) strive to re-establish.

This quest for purity of tests is similar to the insistence of some economists on accepting only pure, market-based tests as the basis for dealing with environmental issues. Which gives us an opportunity to note an important and intriguing aspect of justification theory. Whereas critical/justificatory debates in public favour purity of tests, real-life management systems do not. Just as the firm combines aspects of all six of Boltanski and Thévenot's polities, the management of ecosystems is also a composite of incommensurable orders of worth. Seeking "pure" tests in the purview of one polity is just one part of the dynamics of justification; articulating together different orders of worth, and confronting them to practical situations is just as important.

8.4.3. *ESV: different methodologies rely on different normative frameworks*

This has important consequences for the practice of ESV and its relevance to situations of use in the real decisions. Different valuation methodologies involve different normative frameworks.

Replacement costs (valuing the service on the basis of what it would cost to provide an alternative man-made installation, by comparing the efficiency of various technical solutions, using monetary assessments), clearly belongs to the industrial-market compromise we just discussed.

In contrast, willingness to pay is more market-centred – or could be seen as based on an "opinion-market" compromise. It does not rest on an industrial (production process) substitutability, but on a market one: one kind of good that one would accept to trade for another. Contingent valuation, through willingness to pay, is ready to translate into monetary terms any form of worth. It can be based on aesthetic and feelings (inspiration polity), on functionality (industrial), on opinion (fame), on traditional attachment (domestic). It may be interesting to note how the methodology divides experts. On the one hand, many economists consider it to be the purest form of valuation, because it is untainted by the composite character *e.g.* of replacement costs. It relies solely on the logic of preferences and trade-offs. The difficulty they concentrate on is how to improve valuation techniques – *i.e.* the quality of the test – in a context where important attributes of real markets are missing. On the other hand, many other experts are ill at ease with contingent valuation because it mixes up heterogeneous elements that from a normative standpoint belong to completely disjointed orders of worth. For experts who make efforts to obtain clear and distinct views of the functional values of ecosystems, of their aesthetic values, of their traditional heritage and identity-based values, or who insist on differentiating between opinion-based values and functional values, contingent valuation makes things more confused, rather than clearer.

ESV methods cannot be taken as a set that would rest on the same normative bases. On the contrary, the various possible choices of method(s) imply also a choice of normative framing and foundation. This is an important lesson for the use of ESV. First, because many of the difficulties ESV raises come from normative controversies that could be dealt with much better if they were addressed in their own right by connecting in less generic and more case specific ways practical situations, normative framings and choices of methodology. It is important also in the perspective of using ESV as one approach for assessing worth alongside other, non-economic approaches. In chapter one, we saw how that joint use remains fuzzy and problematic in the literature. One way to improve this situation is through an explicit treatment of the normative bases of ESV, including the diverse forms of normative compromise ESVs rest on, alongside discussion of the normative bases for the other assessments of worth involved in the biodiversity management situation, within a unified framework that would also help linking these two efforts.

8.4.4. *PES as connectors between heterogeneous orders of worth*

If we now turn to payments for ecosystem services, the cases are equally varied. The often heard rationale "better to pay farmers 10 million dollars for reduced water pollution than to pay 100 million for a new water treatment plant" is typical of the industrial-market compromise. The Vittel example provides a good example of the industrial dimension in the transaction. At the core of the issue was the fact that intensified agricultural production systems generated pollution levels that were not compatible in the long term with the processes that produce Vittel water. The solution was to bring farmers to change their production process. Mains steps in implementing the solution were: understanding the social perception by the farmers of their own activity (*e.g.*, the modernity of intensive versus extensive farming techniques); finding an effective alternative farming system through the optimised spreading of composted manure by Agrivair. The "industrial" normative framing of "using efficient production techniques" has been essential in the mostly technical negotiations that led to resolving these difficulties. The monetary compensation component was, as we explained when presenting the case, just one ("market") element in a much more complex situation which also included important elements of "fame" (the reputation of the brand is of strategic importance for a mineral water company) and of the "domestic" polity (as a large company in a very rural environment, Vittel favoured options that preserved well established social relations with the local community, of which farmers are essential members).

In other examples from the South, payments for ecosystem services often consist in providing money to traditional communities so that they can continue with a lifestyle that has side benefits in terms of forest cover, water catchment management or biodiversity. In the negotiation of such deals, domestic, industrial and market orders of worth come into play in field specific configurations.

In real use situations, to be successful from a normative standpoint, each PES scheme has to construct a successful compromise between at least two orders of worth. For instance, between optimisation of drinking water production, preservation of a traditional lifestyle in the watershed, and lost opportunities of making more money. For our analysis of ETB use issues, the important point is that neither money as a language, nor payment *per se* suffice to effect the translation or the connection. These are the result of critical-justificatory dialogue between the parties that may – or may not – reach an agreement that will equate a certain level of payment with a certain level of effort to provide a service. The fact that ESV is hardly ever used to determine levels of payments is an indication that these have to be determined in other ways – that they have to establish equivalences between partly incommensurable values, and that requires the successful confrontation of actors in a negotiation that, alongside interests, is also a negotiation of values.

8.4.5. *Buying land: the normative foundations of goodwill*

Economic tools that consist in buying land or rights for conservation rest on still another configuration. They operate on markets to buy land or rights, but they buy them to extract them from market dynamics. Once bought by a foundation, or by a public entity like the French Conservatoire du Littoral, a piece of land will not be back on the market in the foreseeable future. As we saw in chapter three, once land or rights are bought, some pressures of the market (for instance the pressure for commercial development) are neutralised but the issue remains of managing the land in a viable and appropriate manner. This implies a number of agreements with local users and politicians, with funders and administrative authorities. Such agreements involve the confrontation of very different sets of values (for instance, between the traditional values of some local users, the concern of other actors for local economic development, etc.).

Another conclusion in that chapter was that the whole system of buying land or rights rested on much goodwill and general social acceptance both for extraction from the market and for attribution to conservation purposes. Normative debates are essential for the establishment, or on the contrary for the loss, of such goodwill. It is interesting to examine from this perspective some of the strategies of the Conservatoire du Littoral. (1) At the centre of its mandate is the aim to ensure that the public will retain wide access to the shores of the sea, an access that would otherwise be privatised in various forms through development. The Conservatoire is very careful to organise freedom of public access and monitor the presence and activities of the public on its estate. (2) Since the 1970s, the Conservatoire has commissioned artistic photography on its estate and uses such photography as well as careful design, or commissioned books illustrated by artists, to underline the beauty of the protected coast. (3) In defining the management plan for each property, the Conservatoire negotiates with local elected officials, interest groups and traditional users to conciliate multiple activities on the shoreline. It takes great pride, for example, in proving that there are more farmers on its land after it has bought it than there were before. These three components of the Conservatoire's strategy are all aiming at obtaining support and preventing criticism. They address respectively the normative references of the civic, inspired and domestic polities. (4) But the Conservatoire has also commissioned studies and organised workshops on the contribution of its estate to the local economy. These show for instance that the presence of substantial areas of freely accessible and well managed natural shores increase attractiveness, frequentation and real estate values in the neighbouring communities. So even though buying land extracts it from the market here, this does not mean that the operators are free from critical pressure from the market and industrial polities. There is, as it were, a moral obligation on the Conservatoire to demonstrate that it contributes to the economic well-being of the community, that it can justify its actions in terms of the market polity. (5) Finally, in the internal strategic discussions of the Conservatoire we have participated in, there repeatedly came the worry about a possible press campaign on some badly managed site. The Conservatoire owns hundreds of sites and does not normally manage them directly. There are bound to be places that will be poorly managed for a while, and this could be exploited for a media campaign that could be detrimental to the goodwill the Conservatoire's activity rests on. "Fame", a very positive image for the organisation, plays a very important role too.

Overall, the Conservatoire makes a great effort at justifying its action and preventing criticism. It addresses through specific activities the various orders of worth that may be sources both of critical challenges and of normative strength: the reader will have recognised in sequence the civic, inspiration, domestic, market and opinion orders of worth. More generally, building normative support is needed for the application of any sort of tool in real management and policy situations. A clear reading of that space of critique and justification can help ETB users.

8.4.6. *Biodiversity-banking: a civic-industrial-market compromise*

Finally, offsets and biodiversity banking also have their own normative background. As was discussed, offset is most often based on legal obligation, an obligation that itself originates in the sustained advocacy of environmental civil society groups. It has also strong roots in the industrial polity as its value-equivalences rest on an elaborate system of rational, technical planning of biodiversity. To these two dimensions, the market mechanism of biodiversity banking – being able to buy compensation, rather than having to effect it oneself directly – adds the principle of mutual benefits (for the developer, for the environment, and for the seller of compensation rights) through trade. In our view, its normative background lies in a civic-industrial market compromise: civic commitments lay down principles and create a constraint; science-based technical norms establish precise criteria; trade makes the commitments easier to manage; in return, the civic commitment and constraint generate new business for participants in the biodiversity-banking market.

Biodiversity banking thus rests not only on complex legalities and technicalities, but on a normative construction that is quite sensitive as it combines three different orders of worth. As the tool currently attracts much attention in the biodiversity field, it is striking to see – in publications, in conferences, in the classroom – the level of normative perplexity and challenge it raises. This may reflect the fact that when trying to judge the tool *a priori,* in a very general manner, one cannot formulate a simple normative judgement based on just one order of worth, but one finds oneself playing with the volatile assemblage of three different ones. Maybe this could account also for the observation we have made in conferences and classrooms that there is a great contrast between discussions of the general principles of offset and biodiversity banking, which are very volatile, and discussion of field cases, where the efforts made to accommodate the huge constraints of managing biodiversity in field situations with large development pressures make much more tangible and less inflammatory the confrontation of values in view of practical dilemmas that cannot be reduced to the pure scales of value of any one order of worth.

To conclude this review of the compromises between orders of worth that are involved in the principle and implementation of ETBs, let us note four points. (1)

Justification theory allows a view on the normative foundations of ETBs that is much more precise and nuanced than simply assigning ETBs to the logic of the market, and opposing them to ethical concerns. (2) Different tools have different normative backgrounds, including tools that are often grouped in the same category, as illustrated by our analysis of ESV methods. (3) Different situations of use have different normative stakes. (4) As most pragmatic actions do, using ETBs involves hybrid normative constructions in the midst of pressures for the purification of normative criteria; justification theory helps decipher the complex ways this plays out in practice.

8.5. Is there an ecological order of worth?

Many aspects of environmental issues, disputes and management mechanisms are strikingly enlightened by the framework and concepts introduced by Boltanski and Thévenot's *De la justification*. It is troubling, however, that in the theory the normative foundations for claims relating to ecological issues like biodiversity are systematically to be found in orders of worth that have no particular relation with ecological concerns and values (efficiency, exchange, rule-based governance, etc.). Yet isn't part of the contemporary challenge of biodiversity one of integrating specifically ecological values? As says one of our interviewees: "From our point of view, some of the major issues are about the anthropocentric side of things. If we assume that we are only interested in conserving the things that are useful to us, this is problematic, as there are things that inter-relate in ways we don't understand. Perhaps we haven't realized it yet."

In a way, the apparent difficulty of introducing an order of worth centring on ecology into the normative structures described by justification theory echoes the wider difficulty we obviously have, collectively, to make place for ecological concerns in our decision-making. It also echoes the challenge we discussed in chapter 5, of pursuing at the same time immediate incremental action and action for much deeper changes. This involves working both within prevailing orders of worth, and working on our orders of worth to transform them. To examine the issue, we shall first see how it was approached by authors who used justification theory in the 1990s. We will then turn to our own understanding and proposals.

Shortly after the publication of *De la justification*, in the 1990s, Lafaye and Thévenot (1993), Bruno Latour (1995) and Olivier Godard (2004) raised precisely the question of integrating ecological issues into justification theory.

Although their approaches are in part different, they all underline how, when studying environmental disputes in the field, one sees very clearly how positions on biodiversity conservation or other environmental issues can be justified or attacked on the basis of the six orders of worth introduced by the theory: "How can you envisage destroying a landscape of national fame, which provides an irreplaceable source of inspiration?" "How can you demand a nature reserve here and propose to lock up resources that are essential for the economy and for a community that has used these resources for generations?" "Do you really propose to log this forest for money now and jeopardize the water catchment from which the town has always been able to draw its water?" Etc.

This diffraction (or projection) of biodiversity on incommensurable value scales is quite tangible in debates about ETBs . As we have seen, both within the conservation movement and between conservationists and other actors, arguments use the whole gamut of the major orders of worth that prevail in society in support of their diverse positions. The problem here is that environmental issues end up being mostly advocated on the basis of systems of value that underpin major threats to the environment. This is often underlined by critics of environmental economics (O'Neill, 2007; Kovel, 2002), as they claim that the economic logics underpinning ETBs reinforces an economic culture and an economic system which are at the root of environmental crises. But the same critique applies just as well to any of the established systems of value. The unending quest of more efficiency in the industrial polity ("you can't stop progress"), the reluctance to change traditional values when some of the activities they support become destructive for the environment ("we have hunted this way forever!"), the insistence on implementing public programs that have social support through the political system ("you can't stop subsidising fossil fuel when the citizens say they need these subsidies!"), are examples of the dilemmas that result from environmental causes being advocated based on the same systems of value as environmentally harming development.

This should not lead to discounting the very real contribution of the established "polities" in support of biodiversity. Celebration of the aesthetic and inspirational value of nature (inspiration), approaches based on heritage and on traditions (domestic), anti-pollution devices and eco-conception (industrial), market tools (market), enhancing the visibility of and mobilisation of the public on environmental issues (fame), regulation and appeal to responsibility (civic) all play a major role in dealing with biodiversity issues. For instance the ways in which the ecosystem services approach draws resources from the industrial, the market and the domestic orders of worth to promote biodiversity conservation and restoration can indeed be useful.

But the underlying contradictions it implies beg the question: is it enough? Don't the established orders of worth miss some central values of ecological systems? And in that way, can they not misdirect us into failing in the face of contemporary ecological challenges? Lafaye and Thévenot, Latour and Godard all seem to think that there is indeed such a missing dimension. But they all conclude that "green" values fail to meet the conditions of a legitimate order of worth in the framework of justification theory. Their arguments differ: including future generations breaks the condition of a necessary deliberation on values, the inclusion of non-humans also breaks the condition of common humanity and human dignity as the basis of legitimate orders of worth, the holistic and systemic approach endorsed by environmentalism escapes the tests of deliberation because it overhangs human communities and so claims to dispense with genuine moral debate. Lafaye and Thévenot conclude that green values may lie outside of justification theory's human- and deliberation-based ethics. Latour finds the green polity in the "cosmopolitical" approach we shall discuss in the next chapter. Godard envisions the emergence of the green polity in the gradual consolidation of existing compromises (*i.e.* hybrid normative constructions) that currently address ecological issues, for instance heritage management (*gestion patrimoniale*) or sustainable development.

Our own position (Mermet, 2007*a*) is that the ecological polity does exist and that its central criterion for ordering worth lies in the degree of care taken of natural things, in the degree to which one is ready to make place for natural processes, beings, spaces, in the choices one makes. This is not the place to expound how this hypothesis fits into justification theory. But the prospect of a specific, ecology-based order of worth, leads to an important point about ETBs, their use, and the controversies that surround them. If there is indeed an ecological order of worth, it is still in the process of emerging. Its establishment as a widely accepted order of worth occurs only gradually and under an intense crossfire of attacks against, and defence of, a reordering of values based on ecological concerns. In this process, our hypothesis is that, tentative as they still are, ecology-based values appear not so much directly with the full apparatus of the other polities, but most often as minority allies of the more established orders of worth. What does this mean for ETBs? That beyond all the various configurations of compromise discussed earlier, they also involve an alliance between on the one hand the well-established orders of worth (in particular " industrial" and "market") on which rests their most practical immediate leverage for justification and on the other hand the emerging green order of worth that is most often still not strong enough to stand its own ground alone in critical/justificatory debates.

The result is a high level of ambiguity and tension. In many forms there is an evident concern at the centre of debates about ETBs and their use: are they good or not for the environmental cause? Case after case, the question is raised: is the alliance a fruitful one? Is the concern for natural things (biodiversity, low-management forest, naturally flowing rivers, aquifers, ...) served only in appearance or in reality? Do market or productivity concerns get the lion's share of the outcomes, or does nature get a fair share? The alliance between the emerging green polity and the other established ones is also put to test in the controversies about ETBs. When they are attacked, for instance, because they undermine traditional lifestyles and values by commercialisation of nature, what cause is that attack serving? Is it serving only some rural interest (farmers, foresters, fishermen) and playing against both commercialisation and the interest of taking better care of natural things? Or is the challenge going to deliver concrete benefits to the care of natural things?

Managing the ambiguous alliance between the established orders of value implied in ETBs and an emerging additional order of value based on concerns for nature seems to us an essential issue on the scene of ETB use. Rejecting the alliance completely, as do some critics, is likely to further weaken environmental advocacy and action. Two different configurations may occur. In the first, the critique of ETBs is mostly motivated not so much by ecological concerns as it is by the defence of another alliance with one of the long-established orders of worth (for instance in support of "traditional" rural activities or of industrial rationalisation). But these other alliances should raise exactly the same kind of issues and doubts, in terms of environmental effectiveness, as market-based alliances. In the second type of configuration, a defence of environmental causes that is based only on arguments about the care for nature runs the risk of being overrun by the pressure of forces using market, productivity, tradition or opinion based arguments, so strong is each of them, and so efficient are they at creating synergies when they are challenged by incumbent new values.

An Ariadne's thread to navigate this dilemma can be put in place if one (1) continues to assess outcomes on specifically environmental values, alongside with other values that may be promoted by the use of an economic tool for biodiversity, (2) be wary of supporting critical positions of economic tools if such criticism does more harm than good to the specifically environmental outcomes of their use, (3) refuse to promote economic tools over other sorts of tools (regulatory, participatory, etc.) if the direct benefits are not clear specifically on environmental criteria (and not just on indirect criteria based only on the market order of worth like "sound market mechanisms are more efficient in general and, thus, are better for the environment" or in the industrial order of worth like "our management of ecosystems should be assessed on the level of optimisation and control of the services they provide").

8.6. The "project-based polity": it's not just the tool, it's the way we promote it!

Another and last contribution of justification theory to our examination of the value-basis of ETBs stems from Luc Boltanski and Eve Chiapello's book *The New Spirit of Capitalism* (1999), an effort to discover (or uncover) an additional prevailing order of worth. Based on in-depth research on managerial writings and on the evolution of managerial culture in the 1980s and 1990s, they demonstrate the rapid emergence of a new order of worth based on the capacity to create new connections and operate in temporary networks of action. They name it the "project-based polity". We include it in this chapter because it echoes an important concern about the use of ETBs: doesn't the fact that it so often presents itself in the form of projects or "pilot projects", or "innovative projects", affect the ways in which ETBs are used and the values they promote?

Let us note from the outset, however, that as we address issues of project-based reasoning, it is important to distinguish between two levels of analysis, respectively in terms of organisation, and in terms of values. On the one hand, projects emerge as a major organisational model in all sorts of contexts that extend from the organisation of production in industrial plants to the organisation of public action. On the other hand, and this is what Boltanski and Chiapello study, in parallel with this rise of project-based organisation, a whole set of values has crystallised that sustain the legitimacy of project-based action, and that critiques other forms of organisation.

The central value in a project-based order of worth is a high level of activity, an activity that consists in creating (or acquiring) connections to initiate projects, or to participate in projects initiated by others. A rich portfolio of thriving projects is what makes one "great" in this polity, as it connects the individual and the common interest. Flexibility and commitment, the ability to create mutual interest so that heterogeneous partners will adhere to the project, charisma and attention to others are essential qualities to possess in that order of worth. The roles of facilitator, mediator, strategic broker, partners and connectors are the most valued. Being great in the project-based polity implies travelling light, not being stuck in a comparatively fixed order of things such as provided by the domestic or industrial polities for instance. The capacity to move from one project to the next is a central test of worth here.

Boltanski and Chiapello show in detail how, beyond this rapid characterisation, the project-based order of worth follows the "grammar" of justification theory. They review the reciprocal critiques that the project-based polity holds against the others (for example, against the lack of mobility of persons and the static network of relations in the domestic polity, or the slow-moving technical expertise of the industrial polity and its focus on standardisation, against conditions of the market, such as the impersonal character of the transaction, the standardisation of the product, the transparency of trade that oppose the connection- and trust-based creation of collaboration in networks and projects, etc.). They also devote much attention to the genesis of the project-based polity, which they see as a development from the critical movements of the 1960s and 70s. Both the social and the cultural critique that were at the centre of these movements deeply challenged central aspects of the domestic, industrial, market, civic, reputation-based and inspired orders of values that were felt to stifle freedom, innovation and to lock people in rigid and often unacceptable social situations. The entitlement and capacity to build one's own world, one's own connections and to focus on action for change at all scales – from alternative local lifestyles to project-based strategies for social improvement – are borne from that challenge. Boltanski and Chiapello, however, insist strongly on the fact that the picture is anything but rosy. Born from socially and culturally generous mobilisations, project-based management and the values that underpin it have also led to much suffering in the work place and in society. In a project-based order of worth, those who at one point miss the connections, who do not like to change, or do not have the resources to make others want to collaborate with them are simply left out in contexts where the protections afforded for instance by the domestic, civic or industrial orders have been weakened as the scope of project-based management (public or private) has widened.

Here, however, our interest for Boltanski and Chiapello's project-based polity does not stem from its potential for the social critique of contemporary social conditions, but from the fact that project-based organisation and values pervade the field of ETBs, both in terms of organisation and of underlying values.

From an organisational standpoint, much of the development of ETBs is organised on a project basis. Most striking in this respect is PES implementation. The abundant literature on the subject largely rests on cases of PES projects that manage to connect worlds that had hitherto remained disconnected. Referring to some of the examples we presented in chapter 2, the connection of Cambodian villagers with western bird-loving donors, of Massai with tourists and hoteliers, or of the Guyanese forests with financiers are typical examples of the capacity to create new connections through projects.

Moreover, many such projects are presented as pilot projects. As analysed by Raphaël Billé (2009), these rest on the theory that innovative schemes can upscale and become generalised through a snowballing effect: the success of a pilot project would trigger imitation from other people in similar situations and this diffusion would allow a one-shot project to mutate into a large-scale transformative force. This is far from guaranteed, however. For instance, an innovative scheme can succeed in one case because exceptional conditions are fulfilled in a particular place. In some cases, the innovation is deemed acceptable only because it is not possible to expand beyond it, let alone generalise it. The Vittel case illustrates nicely this sort of situation. Another factor casting doubt is the exceptional amount of effort and commitment that is required by pilot projects. But this may be possible precisely because the project is not perennial, and allows to focus much attention on a small portion of a much wider issue. As Billé summarises it: scaling up from pilot projects may well be disappointing as one moves from high means focused on the most favourable situations to addressing a greater number of harder problems through means that are less focused.

It is important for the community involved in the development of ETBs to be reflexive about the extent to which it is relying on the expectation that innovative experiments with ETBs will snowball. For this, it is necessary first to lay down a lucid diagnostic of the forces at play in the biodiversity problems at hand. If biodiversity loss on a large scale is caused by major structural drivers, it may well be that pilot projects are only able to revert to it in specific conditions and that they do not represent solutions viable at larger scales. A second requirement in order to move beyond blind hope in the snowballing effect is to articulate explicit theories – the kind of action theory that underlies policies and are one of the key concepts of policy evaluation – describing how, through what process, under what conditions, conditional on what means, a pilot project is expected to trigger change. Just relying on the hope for snowballing amounts to an absence of a clear theory of how change is effected and what can favour it or make it difficult or impossible. Third and last, deeper reflection is needed on the values that are underlying the project-based modalities of most ETB development. This is where the problems raised by project-based organisation meet the issues of values, critique and justification.

Many of the project-based values permeating the ETB community are positive. Placing action, "doing something" even when in doubt about success, as a priority overriding the demand for guaranteed effect (at the risk of inertia) has great mobilising value, especially in contexts of large-scale crisis, as is the case with biodiversity. In the same context, trying new connections, connecting the global with the local, testing innovations in management, moving on from what doesn't work, are all essential normative models inspiring action. They should be accompanied, however, by two caveats. First, Boltanski and Chiapello (1999) and others warn, as they analyse in detail the consequences of project-based organisation and thinking, that there is a darker side to the project-based ethos. Aspects of it that concern us here are for instance the risk of opportunistic exploitation of the new connections and of relative indifference to outcomes beyond the – admittedly very limited – scope of each project, or of innovation for the sake of innovation. A second caveat is that the benefits of project-based ethos and organisation (initiative, mobility, innovation, new connections) have to be balanced against the consequences on other orders of worth. If a local conservation project based on a new concept and on temporary funding works perfectly, but undermines the traditional structures, or the administrative management, on which part of the ecosystem management capacity of the community relied, the outcome has to be questioned. Another example from the ETB literature is the risk that paying for services that were hitherto provided for civic reasons is apt to undermine the civic values and thus service provision on a larger scale than a single PES project. The point we want to underline here is that the consequences of ETBs, be it locally or as they may be developed on large scales do not only play out in terms of economics and technical practice, but also in terms of the values that are effectively promoted by ETBs. And such values are contained not just in the design of the tools themselves (as we discussed in detail in this chapter) but also by the way in which we go about introducing them in the field. As the project ethos underlies much of the current effort for innovation in terms of ETBs, it is essential to examine how it interacts with the various orders of worth at play. It can strengthen some through constructive compromise – for instance when a PES project concretely helps a rural community but also confers to it an enhanced degree of legitimacy, as in the example of the Cambodian certified rice scheme described in chapter 2. It can undermine some, for instance when communication on the multiplication of projects and on some successful experiments tends to draw attention away from the civic controversies about the large-scale economic and societal choices that feed drivers of biodiversity loss.

Conclusion

As many authors now insist, solving the biodiversity crisis will require a shift in values. Analysis of biodiversity cases on all scales – from global deforestation to local land-use conflicts for example – shows that there is confusion and volatility of the values at play. One day, biodiversity is tossed around as being worthless and the next day it is hailed as a priceless life-support system. One day it is haggled over against all sorts of interests and wares, the next day it reaches the top of the agenda of administrations and public debate. The heterogeneity of arguments creates a genuine puzzle that is at the crux of biodiversity issues.

Giving more value to ecological concerns certainly is part of the cultural change that is required. But it is essential to realise that this cannot be just a matter of simply adding environmental values on top of the existing system of values. The contemporary situation is one where the coexistence of deeply different and largely contradictory orders of worth – as it were, of too many values already vying for priority – creates particularly complex and moving contexts for attempts to promote new values.

Boltanski and Thévenot's justification theory is a great resource to use as one moves from a simple promotion of environmental values to analysing the value systems and value debates through which actual changes in values can occur and be encouraged. It points to the fact that one cannot – and should not in a democratic society – strive to establish one order of worth that would dominate the scene, but that we all have a capacity to participate in constructing more stable orders of worth and in managing the tensions and compromises between them in real-life practical situations through well-articulated critical-justificatory debate.

This is of crucial importance for ETBs, because value debates are at the heart of their development. (1) Much of the development of ESV, for instance, is driven by the hope to push biodiversity higher in the hierarchy of values used in social and political decisions. This drive dominates all aspects of the TEEB report, for instance, not only through the argument of giving monetary value for balance against economic interests, but through all sorts of practical proposals to augment the attention given to biodiversity in decision-making. (2) However, ETBs do not put in play the whole range of values supporting biodiversity, they also shift the value content of biodiversity by promoting industry- and market-based values. The controversies triggered by these shifts in orders of worth are extremely intense and are an integral part of the social context in which ETBs are implemented. Overall, devoting more attention and more theoretical resources to the treatment of this value-focused dimension of ETB utilisation and development should be a priority for the field.

Let us sum up the contributions of justification theory as discussed in this chapter. (1) It provides a very explicit framework to decipher the crossfire of normative arguments that one is driven into when using ETBs. (2) It helps break away from a simplistic treatment of the issue, that would equate ETBs with the market and oppose it to ethical concerns, as it shows that the market has an ethical dimension too (so that the debate is not between market and ethics but between several ethical orders of which one is market-based). (3) It points to the fact that ETBs rely not just on market-inspired values, but on much more diverse, complex and ambiguous ethical constructions. Understanding the necessity of hybrid normative construction in the midst of pressure for purified normative justification, allowing the analysis of the tensions this generates, illuminates some of the most difficult issues met in the deployment of ETBs. (4) A justification-theory-based discussion of the debatable status of green values versus established orders of worth allows a better navigation of the thin line between necessary alliances and counterproductive capture of ecology-centred values by established orders of worth. (5) Finally, the values immanent in the mostly project-based methods through which ETB activities are carried out should be questioned: on the one hand they foster initiative over fatalism, but on the other hand they may sometimes defeat the very purposes of using ETBs.

Overall, using justification theory to analyse ETB use situations lays down the ground for a more realistic and sophisticated treatment of their ethical dimension. Not only can such an approach provide theoretical and empirical relevance, it has practical value too. Indeed a strong point of justification theory is that it was built from scratch, starting with practical situations where critical/justificatory debate is of the essence. It never loses sight of the fact that as they deal with problems involving issues of justice, actors constantly consider both the principles (orders of worth) and the practical situation at hand, in Boltanski andThévenot's words, justice (*justice*) and relevance (*justesse*). This is precisely the combination that is needed as we advocate a move from confrontations of principle (ETBs as an excellent market tool versus ETBs as contaminating environmental debates with market values) towards putting such claims to the test of actual situations of environmental action and ETB use.

9. Economic tools and innovation: perspectives from actor-network theory

In the previous chapters we presented approaches founded respectively on rule-making (common-pool resource theory) and on the confrontation of incommensurable values (justification theory). We will now turn to another essential dimension of how we manage biodiversity: the nexus formed by our science, our technologies and our social choices is also at the forefront of environmental issues. Currently, this is most obvious in the way we approach climate change. Questions like "What does the science tell us and to what extent do we trust it? Aren't our choices of technology (fossil fuels, renewable, nuclear) at the centre of the matter?" How we make such decisions – about transportation, building, heating, etc. – that may deeply affect lifestyles across the planet is at the epicentre of climate issues. Similar questions, at the same time social, political and technical, are central to the future of biodiversity. Our choices of farming methods, of forestry planning and techniques, our decisions about lifestyle issues, like the amount and the kind of meat we eat, about the fish we eat and where we source it, about energy issues (biofuels, use of forestry biomass, hydroelectricity), etc., are crucial drivers of the biodiversity crisis.

Somehow, however, issues of life-style-connected technological choices seem to be less directly prominent in the biodiversity debate than on climate issues. More in evidence are controversies on which land should be developed (or farmed, or logged), which rivers should be dammed or not and on the level of intensification in land and water exploitation through farming, logging, or fishing. When ESV weighs the benefits of a development project against the corresponding loss of ecosystem services, when PES compensates the maintenance of traditional farming practice or renouncement of logging a forest, when the purchase of land or easements blocks some forms of development or intensification, when an offset program proposes distant ecological restoration to compensate local habitat destruction, they each try to act on local decisions on the allocation of land to production versus conservation, or on production intensification. But whereas such decisions are about who will profit from a given piece of land or a given resource, they also involve choices in technology and life-styles. Issues of resource allocation cannot be cut off from issues in technological choice, lifestyle, and thus politics. So analyses based on resource allocation and rules on values have to be completed by perspectives that give us a precise grasp on issues of technological choice, in connection with the politics of resource allocation, as well as of how we live together.

This is all the more necessary if one considers the magnitude of the changes that our technologies, our level of resources exploitation, our lifestyles (seen from a global perspective) and thus, ecosystems on a global scale are experiencing and will continue to experience in the coming decades. It just cannot suffice to reason on biodiversity issues based only on the attempt to stabilise the situation locally in some key areas, or in others striking some compromise that keeps some biodiversity while sacrificing ever more to the demands of an intensified exploitation of resources. What biodiversity we will have 50 years down the line cannot just be the part that we shall have managed to conserve (although that part is essential); it will also have to be the result of biodiversity redeployments on massive scales. This is all the more true if we consider the effects of global change (climate change, invasive species, etc.) on biodiversity that will put a further constraint on conservation. This redeployment of biodiversity will have to rely essentially on two pillars. The first is a renewed discussion on how we put limits on our exploitation of ecosystems and resources, so as to spare biodiversity. Many national regulation systems, international environmental regimes, the whole protected-areas system are part of that first pillar. The second pillar is an effort to develop and implement production technologies and practice in the management of productive land that have less negative impact or even a positive impact on biodiversity. A good example, involving one of the major biodiversity issues at global scale – how to feed the planet in 2050, with a population of 9 billion without further massive biodiversity loss – is the passionate plea for "ecological intensification" of farming by Michel Griffon (2006). We will return later to the (often very tense) relationship between these two pillars. But wherever one stands when claims for conservation and for redeployment of biodiversity come to clash [31], it is not possible to analyse biodiversity and the tools we can use to manage them without bringing the science-technology-society-politics interface into the equation.

[31] For instance, on whether our efforts in favour of biodiversity in the Congo basin forests should now go towards integrated forest planning or towards a more ambitious program of protected areas.

For this, we will turn here to actor-network theory, which focuses its perspective precisely on this essential nexus. Rather than a unified school of thought, like common-pool resource theory, or one economical and compact theory, like justification theory, actor-network theory is a wide and diverse set of theoretical approaches unified, as it were, by lively debates and by sharing the fundamental view that science, technology and society should no longer be considered separate spheres of expertise and activity but are just intermingled aspects of processes that are at the same time social, scientific and technological. Knowledge, technology and social organisation change together. One only has to think of the deep-reaching changes brought about by the digital revolution, as science, technology and our social organisation and culture change at the same time, to realise the relevance of such a perspective.

In this chapter, we will successively examine two conceptual models from actor-network theory. The first is the sociology of translation model put forward by Michel Callon (1986). This model centres on the innovation process – an innovation that is at the same time scientific, technological and social. The second will be the model proposed by Bruno Latour in *Politics of Nature* (2004) for a political philosophy that would account for how we proceed, as political communities, to take responsibility for environmental issues that were heretofore unmanaged. The two models start from quite different standpoints that complement each other. The first analyses situations from the innovator's perspective, the second, from the perspective of an observer interested in the dynamics of an entire community. Overall, however, they share common foundations and goals, and encourage us to look at science, society, technology, politics as one seamless web of interactions, rather than isolated spheres.

Interestingly, unlike common-pool resources theory and justification theory, the economic dimension and economic tools play no very explicit or leading role in these models. They are present only with supporting roles. So we will have to examine closely what insight such theories, designed primarily to better understand the science-technology-society interface, can bring to the use of economic tools for biodiversity.

Innovation and sociology of translation

It is quite fortunate for us that the provocative and famous paper on "the domestication of scallops and fishermen" through which Michel Callon (1986) introduced his sociology of translation and triggered actor-network theory rests on a case study that falls within our field of interest: the management of a fishery that is threatened with collapse.

The case as analysed by Callon can be summarised in a simple story. Saint-Brieuc bay, in Britanny, is one of three significant French scallop fisheries. At the time of the field study used in Callon's paper, systematic exploitation of the bay's scallops was still recent (about two decades old), but very lucrative and the resource was already decreasing steadily, so that its biology and management made their way onto the actors' agenda. At that point, three fisheries scientists from Britanny discovered during a study trip to Japan how Japanese scallop fisheries are intensified thanks to a technique that boosts scallop reproduction. In a nutshell, scallop reproduction is the result of adult scallops producing swimming larvae that then fixate themselves to transform, after two or three months, into shellfish that will then grow to adult size. The technique consists in installing immersed artificial supports designed so that they will give the larvae increased opportunities to fixate and protection against predators, thus boosting the production of the fishery. When the scientists came back to France, they proposed to test the feasibility of the technique. Scallops in Saint-Brieuc, however, are of a different species than those in Japanese fisheries. Their reproductive biology was then very poorly known. So our three scientists set out to convince research colleagues, funding authorities and leaders of the bay's fishermen's union to support trials in the bay. The first year, trying all sorts of design and location of artificial supports, they obtained a fair degree of success in the fixation of larvae and the experiment was well supported by the stakeholders. In the next two years, however, the experiment was less successful and social support dwindled until an event that Callon describes as the fishermen, on Christmas' Eve, breaking into the tank where the scallops issued from the experiment were kept to be studied. The fishermen made a miraculous fishing session of it. It was to take renewed efforts of better designed experimental supports for larvae, of further negotiations with the fishermen (involving more than just the leaders) and with the authorities, until the technique for fixating scallops' larvae was finally perfected.

Told this way, the story is simple and unremarkable. This simplicity, however, is misleading. It relies on the fact that we know the end of the story. But, as Bruno Latour demonstrates in his book on the sociology of science (1988), there is a world of difference between "done science" and "science in the making". If we move back to the moment when the scientists are starting with their experiments with scallops, nothing is as straightforward as it would seem from reading the story. Are these scallops of a kind that could benefit from provided support for larvae? Are the fishermen ready for anything else than a rush to catching the last scallop in the fishery? Will fishery-science colleagues be ready to support funding for scallop reproduction research? To understand the dynamics of innovation, it is necessary to renounce the false certainties that appear after the event and to adopt a language and a model that fit the state of flux that characterises innovation in the making. For this, Callon proposes three principles. (1) One should avoid using *a priori* sociological models or categories about the actors in the process. It is essential to use the categories that the actors themselves use to characterise the social context of their actions. The social context is a construct of the actors' interactions and should be approached as such, rather than as a projection of the sociologist's *a priori* categories. (2) One should adopt a symmetric view of society and nature, *i.e.* one should not use different concepts, or a different language to describe parts of the process that involve respectively social or natural agencies. In the paper, the scientists are presented as negotiating with the scallops (by offering supports made of different materials, immersed in different places, at different depths, with different designs, etc.) as well as with fishermen or with the funding authorities. The somewhat provocative title of the paper – the domestication of scallops and fishermen – reflects that symmetry. As we will see, this principle is interesting for us as it guarantees the possibility to move seamlessly from one "dimension" of biodiversity management to another and effectively grasp how the scientific, technical, social (and maybe economic) aspects of ecological issues play jointly at all stages. (3) One should ensure free association, *i.e.* renounce introducing presuppositions about the structure of interests, the contour of groups, the behaviour of natural entities, etc. This will allow a better, detailed monitoring of how the (human or non-human) agents in the innovation process define each other and constantly redefine themselves, how their alliances and associations are changed again and again. Again, it is only after the fact that the stakes and outcomes of an innovative experiment are clear. During the process, the stakes, the agencies, the outcomes are all highly subject to doubt and the way these doubts are managed through the various stages of the process should be the centre of attention – an attention that could not be obtained – if the doubt experienced by the actors were replaced by artificial certainties from the analyst. The actors are shaped by actions and connections, which the phrase actor-network theory conveys, suggesting that it is the network, *i.e.* the (re)connecting process itself, that is the primary mover of change.

How does using this principle help capture the doubt and instability experienced in the process of innovating scallop reproduction, and the way the involved actors managed to transform them into a new, stable system – at the same time scientific, technical and social – of managing the fishery? Callon adopts as central the point of view of the scientists. They are seen as the main agents of change. In terms of the paradigms of action in chapter six, this model clearly belongs to the minority action for change paradigm, with the innovator as the actor driving change. What do these innovators do? To build the new connections that will effect a change from the initial situation of doubt to the final situation where the innovation pertains, they will have to move through four successive stages of a process that Callon calls "translation" which consists in renegotiating identities, interests and relationships amongst all agencies (human or non-human) involved. The central stake of all four stages is for the innovator to establish his innovation as an "obligatory passage" which the various (human and non-human) actors will be brought to adopt as the new avenue to reach what they are aiming for.

The first stage, Callon names "problematization". The innovator lays down the obligatory passage as a hypothesis: if the various actors at play (again, human or non-human) were to take that passage rather than just continue as they do now, they would all be better off in the end. In the case of the scallop fishery, if the larvae fixate themselves on supports provided by the researchers which will protect them from predators, they will have a better chance to live; if the fishermen support the experiment, they have a better chance to continue drawing a revenue from the fishery, etc. It is already clear at this stage that if the innovation works, it will not be just the same situation with the innovation added, but a situation that will have been transformed in some depth – here, from a free-access dwindling fishery to a technically intensified fishery and all that this implies in terms of organisation.

At the second stage, benefit-sharing promise (*intéressement*), one has to move from hypothetical problematization by the innovator to an actual expression of interest by the actors that would be set in motion by the innovation. In the scallop case study, the authorities will have to grant research funding, the fishermen to let the scientists install their devices in the bay, and the scallops, to fixate (if they so do) on the devices. At this stage new connections are actually established, new links between scallops larvae, scientists, fishermen and funding authorities, who had no previous connections but are becoming jointly part of a new tentative management system. To create a new connection one has to relinquish or break some of the connections associated with the former ecosystem or system of management – this will prove to be an important point when we shall use Callon's model to envisage ETBs as innovations.

The third stage Callon names "enrolment". This is the actual test for the entities participating in it and a new round of intense negotiations. Regarding the scallops, the scientists are ready for any concession: what kind of support do they prefer? How deeply immersed? Etc. And with regard to their academic colleagues, our scientists spare no effort in trying to convince them that the first results of the trials do indeed prove that the species of scallops in Saint-Brieuc actually goes through a phase of larvae fixation on solid supports. It is worth underlining that through the problematization and enrolment phases, all sorts of strategies can be used. As Callon writes:

"anything goes". This is clearly an important element in the instability and unpredictability of innovation situations: we do not know if the obligatory passage will work, nor do we know what exactly will have to change for it to work.

At the final stage – "the mobilization of allies" – the test moves from the reduced number of participants in an experiment or a pilot project to the much larger number of actors that will have to take the obligatory passage if it is to make a full-scale difference. The question becomes: to what extent were participants in the experiment representative of their class? Will the mass of scallop larvae behave like the few hundred fixated on the experimental supports? Will the fishermen adhere to a system that was supported by their leaders, but that, to succeed, will have to break away from the lack of constraints and organisation that presided over the two decades of the fishery? An important point here is that whether or not participants in the test were representative of their class is not really predictable, but is established as a result of putting to test that representativity by moving from a small-scale experiment to full-scale practice.

There is a world of difference between just a hypothesis on equivalences and actual, concrete equivalence that will work out in the field. Callon uses the term translation for the entire four-stage process that leads from the former to the latter, because its operating principle is establishing equivalence between perspectives (vocabularies, actions, stakes, etc.) that are initially incommensurable. Scallops, fishermen, academic institutions and funding local authorities do not pursue the same things nor do they speak comparable languages. But through the translation process, equivalence is successfully established – in Callon's example, between the fixation of larvae, more sustainable and higher income to the fishery, better understanding of that particular species of scallop's reproductive biology and a contribution to local economic development. And once such a translation has gone through the stages of testing and been successful, it becomes the new natural condition and set of relations of the place, a new socio-ecosystem that will rapidly acquire a form of obviousness (about the actors, the relationships, the techniques, the underlying knowledge) such that we will forget the long process of flux that led to it.

9.1. ETBs in the light of Callon's "translation" model

The fit between Callon's translation model and the kind of problems we face as we try to make use of ETBs to solve biodiversity issues is essentially twofold.

First, the aim of ETBs – as of any biodiversity management tools – is to help reconfigure relationships between human and non-human actors, including the technological devices and practice that are inseparable from how we manage land and resources as well as from our lifestyles. Not all ETBs, however, involve technological innovation in any direct or explicit manner. Some do, for instance, when a biodiversity banking program rests on innovative ecological engineering for ecosystem restoration. But in many cases, the innovation is rather of a procedural and organisational nature. The whole point of economic tools is to provide specific levers to operate changes leading to new arrangements between people, and between people and ecological systems. The central stake of ETB use is to find out to what extent, and under what conditions ETBs can really deliver on this promise.

Second, ETBs are to a large extent about establishing equivalents. Valuation of ecosystem services proposes equivalents in monetary terms between ecosystem functions and human needs. Payments for ecosystem services concretely operate an equivalence between ecosystem management by the providers and the service as valued by the user. As for biodiversity banking, the very concept of compensating impacts puts the question of equivalence at the centre of practice and debate.

In short, in the light of Callon's translation theory, ETBs are about providing leverage to reconfigure situations involving people and nature, by establishing specific forms of equivalence. This perspective can help ETB users in two quite different ways.

The first, closer to Callon's original sociology of innovation point of view, is by allowing a less prejudiced, more accurate and precise analysis of what effectively happens when one uses an economic tool to try and change a situation in the field. The main focus of the translation model is to follow how the actors themselves, in the flow of action, effectively translate (establish equivalents) between entities, attributes, desires, concerns, that are very heterogeneous and largely irreducible to one another. This requires that one let go of pre-conceptions of how people are supposed to reason and operate, so as to be able to observe on the spot, in real time, in each specific case, how they actually reason and operate. Anyone familiar with the practical and theoretical debates about ETBs can form his own judgement as to the number of stereotypes – ranging from theoretical models of how actors choose and act to the clichés brandished by fans and detractors of economic tools – that are at play. Such stereotypes clutter up the discussion space that could be used to examine actual practice and relevant theories of use. The translation model calls for a suspension of judgement until one has actually observed and documented how people in fact establish equivalents and make their deals in a given case. Beyond this welcome suspension of judgement, the translation model provides concepts that can be quite useful to guide observation of how people (re)negotiate their connections between themselves and with technological and natural entities.

Observation, however, is not that directly useful for the ETB user engaged in action. For him, the central question is how to get his innovation adopted so as to make a difference in the field. Here comes a second possible use of the translation models. The four stages of translation – problematization, benefit-sharing promise, enrolment and mobilisation of allies – describe in a very relevant way the successive challenges that the innovator will have to take up successfully. Admittedly, they do not indicate how successful problematization, etc., can be achieved. This is because what can work is context-dependent to the extent that there are no general recipes for success in ETB use. More important is the ability not to stray off course as one searches in the specifics of each use situation the heterogeneous elements that can make a tool work for a positive transformation. Asking oneself again and again relevant questions is of the essence here. We think that the questions that constitute the translation model – How do I see my tools as advantageous to the situation's actors? How do I get them interested? How do I get the first steps of concrete implementation? How do I get them to stick with the program? – are a good guide for ETB users. They may seem simple, but the logic of network reconfiguration that underlies them provides them with depth and breadth. Furthermore, they also lead to bridging the gap between action and observation. If an ETB user wants to effectively involve others in the implementation of his tools, encouraging him to observe how they really assess situations and how they make their decisions – rather than presupposing that he knows how they do it – is a very good piece of advice indeed. So although the observer's and the innovator's situation are very different, the connection between the two is very relevant indeed.

Let us now examine in a translation model perspective the four types of ETBs we reviewed in the first four chapters of the book.

9.1.1. *Ecosystem services valuation: new Esperanto, or operative translation?*

As we first turn to ecosystem services valuation (ESV), the ambition of providing a tool for translation is immediately apparent. The logic that founds ESV can be summarised as a chain of equivalences from (a) ecosystem structures and biodiversity to (b) ecosystem functions to (c) ecosystem services to society, to (d) value for society, for which (e) monetary evaluation can establish equivalences with all sorts of heterogeneous other social values. ESV proposes to act as a translator in a literal sense, as it puts forward a new, economic language that could become common to all the various spheres that are connected in this chain of reasoning. The economics of ESV are about how that language works, and about how values in heterogeneous spheres (for instance, primary production in a wetland, or the demand for recreation in ancient forests can be translated into the language of economics). The hypothesis behind the promotion of ESV for the treatment of biodiversity issues is that providing such a common language will facilitate concrete new connections between the various spheres involved, so that for instance considerations on the needs of healthy ecosystems will lead to effective changes in human actions that impact them. As we showed in chapter one, however, although we have been very active for decades in so translating ecological issues into economic language, such translations (ESV) have not had nearly the effects that were expected in terms of changing decisions.

Has there been a problem in the translation? At this stage of the discussion, we must remark that we just used rather indistinctly two meanings of the word "translation". The first is the commonly used linguistic meaning of turning from one language to another, of making a statement easier to understand for someone. This is evident for example in the assumption that decision-makers would understand ecological issues better if they were translated through ESV into a language they are supposed to understand and trust: with money as its vocabulary, and economics as its grammar. But the translation concept used by Callon is quite different from just a linguistic translation. It involves various (human and non-human) actors concretely reorienting their reasoning and action in ways that will actuate new connections. This meaning is based on another common use of the word "translation" as it means to change the form, condition or nature of something, as in the example: "to translate words into deeds" – or, we may propose, "translating economic valuation of ecosystem services into actual changes in the way we manage ecosystems".

If ESV is to make an actual difference, it cannot just propose translations into economic language and hope that this linguistic trick will effect a real transformation. What is needed is an operative translation in the concrete sense of actuating new conditions. Callon's "translation sociology" helps us grasp what is involved for ESV to be used to that effect.

(a) ESV has to become an "obligatory passage" in the sense that all actors involved do actually feel that they would be better off guiding their understanding of situations and their choices on it, than they would be without ESV.

(b) This also involves that they be ready to break some of the old connections, *i.e.* to relinquish some of the ways they make decisions now, to replace them with an ESV-based guidance. If we think about firms exploiting natural resources guided on tangible profitability or about political leaders who decide based on a complex combination of negotiating with stakeholders and political forces and of nurturing their GDP and employment figures, the challenge is immediately apparent. Translation is not about just adding one layer of language to the system, but about bringing changes in the configuration of how things are done and doing away with some of the old patterns.

(c) This can be effected by innovators engaging in a relentless series of negotiations with all involved, negotiations in which they will all relinquish something in exchange for something else that is brought by the innovation. It is important to note that this is in no way specific to ESV. If we look at economic figures that are actually used daily for political decision-making (cost-of-living indexes, or official inflation figures, for instance), they are definitely not just the product of academics or experts isolated from political life. They are the object of intense controversy, lobbying, expertise and counter-expertise, negotiation and rule-making that confer them the legitimacy that makes them usable as political decision-making tools.

Why would things work differently for ESV? Valuations start becoming possible tools for decision-making when they start to be the object of serious social involvement and political negotiation – about the methodology, about the data, about how and by whom results are to be presented and disseminated. Participatory ESV makes one step in that direction by involving social actors in the valuation process. But it is only one step, in that to be a robust political decision-making tool, there has to be a sufficient level of political involvement, *i.e.* those engaging in the valuation have to be credible political forces, and political institutions have to confer to the valuation process and results a measure of political legitimacy. (One could transpose the same reasoning to any tool used for decision-making in the firm: credible managerial forces have to be involved and a sufficient measure of legitimacy and expected impact has to be conferred by management). From the inception of our interest in ESV, we have been struck repeatedly by the fact that most of the instances of actual use of decision-making are in contexts with a strong legal dimension – such as trials where ESV is used for determining compensation payments, or permitting procedures for development where valuation is requested as one of the conditions prior to permitting. In such instances, the two conditions we just underlined are fulfilled: there is indeed actual contradictory involvement of the major players in the valuation, and there is an institutional value placed on the valuation process and result. In such instances, ESV has indeed become an "obligatory passage", leaving few options to participants other than to participate in valuation (through expertise or counter-expertise) or let others decide for them.

(d) The more detailed description of the translation process as proposed by Callon may provide a useful model to help follow the process through which an ESV may eventually create new equivalents and new connections – *i.e.* get used with effect. The problematization stage focuses observation on relevance: how explicit are the hypotheses on each link in the chain of equivalence that seeks, through a given ESV, to connect an ecosystem and social values? How credible are, *a priori,* those hypotheses? The benefit-sharing promise (*intéressement*) stage raises the question of comparing, from the point of view of each of the (human or non-human) actors involved, between accepting the ESV and its results as an obligatory passage and sticking to its former ways. As we saw in chapter one, this comparison requires that we understand clearly and in detail – *i.e.* much better than we usually do today in most ESV studies – how actors really make their decisions. The concept of benefit-sharing promise makes the stake of such analysis particularly clear. If we hope actors will prefer ESV as the way to analyse ecological issues, understanding how they carry out that analysis (even implicitly) without ESV is not an option but... compulsory! The enrolment stage directs our attention to experiments and projects where ESV, even at small scale, actually works, that is, is really used to determine a change of trajectory in management. As we have shown elsewhere (Laurans *et al.,* 2013), such studies are quite rare. Finally the last stage, mobilisation of allies, points to the issues that separate an experiment from a robust mechanism that will repeatedly elicit the expected responses from the entities it targets. As we look at the generalisation potential of a pilot study, are the characteristics that led the actors to abide by the compulsory use of ESV very specific, or are they sufficiently general to hope that the process will work in other contexts? What defines representativeness of the context of an ESV experiment, *i.e.* the fact that if it has worked (if it effected tangible changes in management) in a given pilot experiment, it may be expected to work in other, similar contexts too?

To sum up, in chapter one, we have shown the seriousness of the present deficit on actual use of ESV and the need to understand much better the decision-making a process which one would like to transform by making ESV a significant input. Callon's translation model does provide an interesting analytical framework to follow such process without artificially separating the multiple dimensions of decisions about ecological issues. It provides us with clear and detailed concepts – like translation and its four stages – to guide action and to make the difference between what is just an abstract equivalence in principle, and the kind of equivalents that can realign reasoning and behaviour. For ESV, the question is indeed to see if it is bound to remain mostly a universal language, but with very little hold on practical dealings – a form of Esperanto – or whether it can become an element of support for actual innovation and change.

9.1.2. *Payment for ecosystem services: when do the new connections it operates actually work?*

Payments for ecosystem services are not a technical innovation, but they may be considered, in some contexts as a form of organisational innovation: a new connection between ecological processes and a set of heterogeneous social actors. Callon's translation model easily fits the needs of analysing the creation of such connections. If it is to work, a PES scheme has to become the "obligatory passage" through which all involved actors will rechannel part of their business. Take a payment for late mowing of wet meadows to protect corncrakes. Is it the best available way for environmental organisations to obtain changes in farming practices? Is it compatible with the farmer's production system and is it advantageous enough for him to take the trouble of changing practices? Are the vegetation of the meadow and the corncrakes themselves going to respond to the change in the expected way?

Applying Callon's model to analyse the use of a PES seems to us quite straightforward. Many components of the model should be quite useful – for instance, the interest of examining what connections have to be broken or loosened, to create the new connections promoted by the PES, or the various stages leading from the concept to wide and robust adherence, or again, the importance of considering what is in the deal for non-human entities.

Let us just underline two benefits of the model. The first is that it allows to 'follow' decision-shaping processes as they move fluidly across dimensions that are too often disjointed. If we wonder what will make the PES sufficiently attractive for the farmer, all kinds of considerations will have to come into account: production technique, equipment, the image he has of himself as a professional (is it based on maximum production, or on an integrated rural lifestyle and land management), the impact on his income, the possible risks of harvest loss, the administrative pressure on the farm, the opinion of neighbouring farmers, etc. The second is that it is completely open to any sort of local factors and configurations, as the analyst is not applying pre-established categories, but is rather following the way the actors themselves analyse and categorise the contexts of their action. The capacity to reconnect the economic tool with the multifaceted trade in which it is to be used was precisely what we found out to be much needed, in our analysis of the PES literature in chapter two, and the four-stage translation conceptual model can help towards this reconnection.

9.1.3. *Buying land or rights for conservation*

Looking at the buying of land or land-based rights for conservation, the analogy that springs to mind is private property as an obligatory passage. That concept captures in an interesting way the striking contrast we noted in chapter three on the action situation before the purchase and after the purchase.

Before the purchase, the need for goodwill dominates: the proposal to buy has to go through all the stages that will lead from the initial idea that land-rights purchase may be a solution to the actual transaction – including multiple transactions if, as is often the case, one has to buy from a great number of owners over a considerable extent of time. One might write that it is not only that the operator has to buy the land (or easement), but that he has to convince many different stakeholders to buy into the scheme. This relies on these stakeholders seeing the benefit for themselves of enrolling in the scheme (by selling, by funding the purchase, by authorising if needed, by granting tax-breaks or special planning rules, etc.). The stages one has to go through as one drives up to the purchase are similar to those in Callon's translation model.

After the purchase, the owner is in a position where his accord is an obligatory passage for about anything anyone would like to do on the land that would touch on his rights. The interesting point is that this does not lock the land in an impoverished configuration of use. Many cases of organisations that own land for conservation show that they are involved in re-negotiating with all sorts of users. By blocking some irreversible and massive land-use changes (intensive agriculture, plantation forestry, urbanisation), the constraint established by for-conservation-ownership opens the possibility for many other uses of the land that would have been suppressed if it had been left to repeated property transactions on the market. The central teaching of the translation model here may be that an obligatory passage can be a powerful lever for change, innovation and adaptation. Discourses that oppose "hard" conservation measures (protected areas, land & rights purchase) on the one hand and integrated management on the other are simply discounting the fact that spatial differentiation of use is an integral part of integrated management. Can one contemplate an ecological city without green areas? And can such green areas exist without some combination of planning and/or private property that stops people from developing them? In the same way, integrated coastal management requires that there remain areas of comparatively natural, undeveloped shoreline. And under high pressure, these have to sit on "hard" measures. Especially in situations where biodiversity is under a lot of pressure, the translation model's dialectics between an obligatory passage, innovation and negotiations to persuade others seems to us a more useful guide than stereotypes opposing regulatory, economic and collaborative approaches.

9.1.4. *Offsets and biodiversity banking: a tool for reconfiguration of the land*

Finally, as we consider offsets and biodiversity banking, we again find that the concept of an obligatory passage captures quite well the combination of rigidity (in the ecosystem loss cap, in the enforcement of it, etc.) and of flexibility in the spatial redesign of the land and of ecological landscapes, in particular through ecological restoration – a combination of rigidity and flexibility which is at the crux of offset and biodiversity banking. The cases we presented in chapter four illustrate some of these points. In the Australian biodiversity banking program, developers do have the option not to enter the scheme... provided they accept a lengthy and very constraining authorisation procedure. This is "*intéressement*" at its best: open a new way that is comparatively more attractive than the other ways, and you may obtain significant reconfigurations in the decision-making and ecological management system.

The focus of Callon's translation model on innovation sheds light on an important issue in the debate about biodiversity banking that, in our opinion, has not been underlined adequately by the literature so far. Much of the debate on, and the critique of, biodiversity banking bears on those connections that are lost through trading. O'Neill's critique of biodiversity banking bears on the absence of absolute ecological equivalence, on the sense of place that will be lost, on the loss of neighbourhood connections that people had with nature and that will be traded away through the system. For their part, defenders of biodiversity banking tend to argue that something is better than nothing: if a development is going to break some connections anyway, by creating ecological damage, by generating environmental impacts people are going to suffer from, then it is better that new ecological connections be created somewhere else, rather than none.

Set up in this way, the controversy misses the perspective of the potential of the system for social-ecological innovation. Considering the magnitude of the changes pending in our use of land and in the biodiversity management associated with it, piecemeal substitution moves for conservation can only be part of the picture. The bigger picture has to involve large-scale redesign of landscapes – a redesign that must include conservation, but not be restricted to it – and it is also as a tool for this redesign that offset and biodiversity banking have to be considered as providing a lever for new connections and innovation. Here, the translation model again points to the potential usefulness of a relevant obligatory passage. But more importantly, it reframes the problem as one of reconfiguring social-technical-ecological connections. Through its focus on the innovators position, it does not try to posit the issue directly on a large scale, as planning approaches do, but points to the potential of local obligatory passages to be instrumental in operating larger scale reconfigurations. This is precisely the perspective we think is needed to push beyond the current state of the biodiversity-banking debate, as it may guide operators and analysts to ask: "How useful is this biodiversity banking scheme as a lever for what larger scale reconfiguration of social-ecological connections?".

9.1.5. *Negotiating innovations or institutions?*

Before we turn to Latour's Politics of Nature, which addresses more directly this larger scale of science-society-nature recomposition, we owe the reader an epilogue about the Baie de Saint Brieuc scallop fishery.

Based on reports of fisheries scientists , the fishery has been stabilised for now. The major factor here has been increasingly severe limitations on the fishing effort, that have reduced it by a factor of 7 to 9 since the 1970s. Techniques to boost scallop reproduction are not used in the fishery. But the technique of larvae fixation does play a role. In the tight technical-regulatory management of the fishery, it provides important information for the assessment and anticipation of stocks and yields, and thus for the ongoing negotiation on fishing effort rules.

This epilogue allows us to conclude with three remarks.

The first just points to a possible misunderstanding of the translation model. It is not just about innovation *per se,* but about the way innovation – technical or not – is part of the seamless web of relations that constitute a biodiversity management situation. Who operates the scallop larvae fixation system to provide the data (the local fishermen's association or the scientists?). How do they proceed exactly? How are the data integrated into stock assessment? And how does that assessment translate into management decisions? These questions are central to the fishery's management. They connect science, technical issues of the fishery and rule-making so tightly into one that it would make little sense now to analyse the one apart from the other.

The second remark segues on our introduction to Callon's paper: this would have been a typical case for analysis using common-pool resources theory. The epilogue reinforces that impression by showing the pivotal role of the ongoing negotiation of the set of rules governing the fishery. Moreover, Callon's 1986 paper makes no reference at all to the rules-based management of the fishery, whereas the fishermen, fish merchants and the authorities had been engaged since the mid-1960s in a regulatory-based management that was sometimes tense enough to make media headlines. Callon's exclusive focus on the dynamics of innovation at play in the case is of course not objectionable in the context of proposing the translation model to enlighten this dimension of management issues precisely. But it would be an entirely different story if one were to start from innovation as an important dimension of biodiversity management and from there move to advocating that we should foster innovation rather than rule-making, and have more innovation and less rules. As we have tried to show when applying the translation model to various sorts of ETBs, obligatory passages (both technical and institutional) are the fulcrum of innovation. The question is not so much "How much obligation is there?", as "Are the obligations set in the right place?". The latter question can only be answered by considering jointly the obligations and the innovation dynamics at play. Another concept of the translation model concurs in that direction: the idea that as the network of relations transforms itself, as new connections and "obligatory passages" are put in place, one has to cut some of the previously existing links. Creating new connections through ETBs cannot be separated from cutting some of the old connections. For instance, it makes little sense to create PES-like subsidies for biodiversity friendly farming if, at the same time, one does not decrease the level of "brown" subsidies that encourage the ongoing expansion of biodiversity damaging farming systems and practices.

A third and final remark on the application of Callon's translation model to ETB use is that in Callon's case study – and this is the case through most of the actor-network literature – economic aspects are present indeed (the scallop fishermen are clearly driven by their own economic interest), but they do not play the leading role in the process, nor do they seem to operate following their own, independent logic. They rather operate jointly with all sorts of other aspects that go from biology to politics, forming specific combinations the logics of which change along the process. This may represent an extension of the foundational motivation of actor-network theory, *i.e.* breaking traditional separation between science, technology and society, to breaking similar separations with economics. If science, technology and society are to be treated as facets of a seamless process, economy could and should be treated in a similar way, as one facet, or aspect, or moment, of a process of reconfiguration that touches all dimensions of socio-ecosystems. To do so, one needs to adopt a conceptual model that focuses on these transversal connections, and as we have shown, this is precisely what Callon's translation model offers.

9.2. "Politics of Nature": a political philosophy for ecological issues

Michel Callon's translation sociology aims at understanding the innovation process at a "micro" level, from the innovators viewpoint, as it were. For his part, Bruno Latour (2004) developed over the 1990s a theoretical framework for a political philosophy of ecological issues that adopts a "macro" perspective, looking at how the resolution of ecological issues changes connections and relationships from the point of view of the whole community involved.

To introduce Latour's "Politics of Nature" framework, it may be best to start from the overbearing concern and motivation that is expressed throughout the book: the rejection of scientism. For Latour, the usual views on the relationship between science and policy amount to an excessive delegation of political decision-making to science and lead to a depoliticisation that is highly detrimental to our capacity to resolve environmental issues. Let us just note here in passing that it is precisely a similar kind of depoliticisation that is often reproached to economic tools for biodiversity. But before focusing on ETBs, we need to present the "Politics of Nature" framework.

Latour compares the usual view of the science and policy-making interface to a dual assembly parliamentary system. One assembly is science, which debates on facts and reaches a decision about what is or is not the case, about causes of problems, about technical conditions for resolution, etc. The second assembly is politics, in whose arenas values are debated: principles for political decision-making, preferences, the interest of various groups, etc. In this traditional perspective, the workflow for making decisions about ecological issues is simple: politics raises an issue, science determines what solutions are available, politics chooses amongst the solutions based on a deliberation about the values and interests at stake. The problem with this distribution of powers is that science is in a position to trump politics. In its own deliberations, it includes choices with major political consequences (for instance in the ways it establishes research agendas, deals with uncertainty, chooses which variables to include or not in studies, etc.), so that when it delivers "facts" and "options" for politics to decide, there is often really no room left for deciding. We may notice here that the problem raised by Latour is similar to the analysis that led Jerry Ravetz (1986) to introduce the notion of "post-normal science", *i.e.* the fact that if we are to make hard decisions on the basis of soft science (think of the uncertainties in climate models), then we need to participate in the science-making process itself, because much of the political decision will be foreshadowed in the science. Latour's subsequent treatment of the issue is different, however.

For him, Nature as a concept is the fulcrum that provides the leverage for science to dispossess politics of its grip on things: about what is "natural", we can do nothing but what the scientists tell us we can do. As for society, if it is seen as composed of humans only, without the things that their life depends on, it becomes an assembly cut off from reality – a critique that could refer for instance to those forms of environmental sociology that reason as if environmental issues were just "representations" in people's heads and discourse. Latour insists that we live in a seamless hybrid world of things, man-made or not, of people and of relations that connect people to people, people to things and through things to more things and people, things to people and through them to other things, etc. The problems we have with ecosystems are issues within this fabric of people and things, not problems that "society" would have with a "nature" external to it.

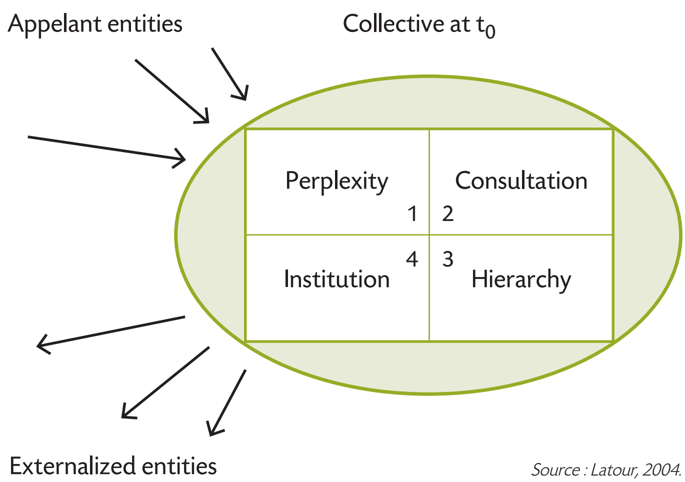
To get rid of the duality between nature and society – and thus of the duality of science and politics and the latent domination of the former – Latour proposes to consider that we are all (people and things) members of a collective. Politics is then the self-management by this collective of its internal affairs, affairs that concern and are managed jointly by people, other living beings and things. Things do interfere in human affairs, for instance through ecological catastrophes that change human lives. In a former research on wetland restoration on the French coast (Laurans, 2000*a*), we were struck to find a similar three-stage sequence in each of three case studies.

First, it took several years to painfully negotiate between local actors with diverging interests a restoration program targeting a high level of control on the ecosystem, along with a management plan. Then a natural event (a storm, an epizootic, etc.) intervened that physically did away with that restoration and management. Finally, this forced the actors back to the table, and led them to adopt a new, much more open and realistic, restoration and management plan. These politics that are made at the same time by humans and things Latour proposes to call "cosmopolitics", but for simplicity's sake, we shall just use "politics" from here on, taking it as meaning the politics of a collective of both humans and non-humans.

This perspective leads to a renewed concept of environmental issues. In the "Politics of Nature" framework, an environmental issue is an entity (a species on the brink of extinction, a molecule making water unfit for consumption, a hole in the ozone layer, etc.) that has no place yet in the order and political life of the collective: its needs or demands or threats are not dealt with politically. The whole issue then is to determine whether and how the collective should expand by making place for the new, problematic entity in its political order. Such issues are part of a wider process of expansion of the collective. New people who arrive, new technologies that are introduced, new knowledge that is obtained, phenomena that disrupt (storms) or create opportunities (the return of a nationally extinct species like the wolf in France) constantly lead us to revise the perimeter of the collective and its order. In this framework there is no fundamental difference between analysing how we make place for internet-linked technology and changes, and how we make place for ecological entities. On each issue, the problem is to determine who (and what) is and who (and what) is not going to be part of the political order of the collective, and what relationships will have to change if a new member is to be included.

To analyse the political process that will operate this determination, Latour proposes a model involving two tiers of deliberation. The first, "recognition", is about deciding on the principle of inclusion or not of the entity into the collective. It addresses the question: "Are we prepared to deal with this problem (or opportunity) or should we continue with the way we do things now?" The second, "ordering", is about deciding on the kind of reordering through which that inclusion is going to be effected. It addresses the question: "Since we have decided to deal with this problem (or seize this opportunity), we must make changes, with winners and losers; what changes are we going to make while still remaining a politically viable collective?"

Figure 10 *The collective defined by its movement*



Source : Latour, 2004.

Within each of the two tiers, Latour distinguishes two stages, leading overall to a four-stage model of the process that determines inclusion or exclusion of an entity (for our purposes, an entity raising a biodiversity issue) into the collective. Each of the stages materialises a different, fundamental principle. Let us examine them in sequence.

Perplexity describes the state of confusion, scepticism and alarm that accompanies the first stage of our realising that something might constitute an environmental problem. Such nascent environmental problems are easily felt to be yet one more entity trying to force its way into our political, social and economic life. The problem is, that this life such as it is, is already more than complicated enough. It is only natural that many of us are not eager to welcome new issues on the agenda, even if some others feel strongly about them. The principle that should govern that stage is, in the imperative formulation chosen by Latour: "Thou shalt not simplify the number of propositions to be taken into account in the discussion" (*ibid*, p. 151). In other words: if we resist taking new issues seriously too much, the collective is at risk of cutting itself off from reality. The comfortable sense of solid reality that develops when we stick to the familiar connections is misleading: it has to be challenged by accepting to be perplexed by new threats and possibilities.

Consultation is the next stage. If an interval of perplexity leads us to think that there is indeed an issue that we shouldn't push aside, the question arises of how this is going to affect the members of the collective. In any political community, each member needs to examine how an issue affects him before proceeding into a deliberation on joint action. The underlying principle: "Thou shalt make sure that the number of voices that participate in articulating proposals has not been arbitrarily curtailed" (*ibid*, p. 154). Again, such "voices" are not only human voices. Consultation also has to involve the other living beings and things that are part of the collective: have the impacts on them been given consideration in the deliberation?

Prioritisation is the task that follows consultation. To make place for the new entity (a new environmental problem, a new technology, etc.) into the collective, some of the existing connections will have to change or be broken. The central stake here is to do so without compromising the existence and identity of the collective. In imperative terms: "Thou shalt discuss the compatibility of new proposals with those that are already instituted, so as to keep them all within one, common world that will give them their legitimate place." (*ibid*, p.155). This is a phase of intense negotiation, such as dominate the scene once an environmental problem has been widely recognised after years of debate, but action is not yet up to the challenge. The stake and the difficulty here is one of moving from diagnostic to actual change – a change that affects simultaneously political and economic relations, technology, worldviews.

Institution closes the cycle. After doubt, debate and negotiations have run their course and the changes are being made, they need to be officially recognised as the new reality (social, legal, technological, scientific – again, they all go together) of the collective. The principle: "Once the proposals have been instituted, thou shalt not discuss the legitimacy of their presence in the life of the collective." (*ibid*, p. 152). The rationale is that in any political collective, new issues are arising all the time. It cannot do to indefinitely rehash those that have already been dealt with. To move on, they have to become the new baseline for the life of the collective, as it is getting ready for new challenges that will inevitably arise. This is just as true for issues that were formerly categorised as "political" (for instance through law) as for those that were seen as "scientific" (the acceptation of something as fact by the scientific community). In contemporary environmental issues – just think of IPCC and the climate negotiations – issues of science, technology, economics and politics are interlinked in such a way that it is increasingly futile to claim that they can be neatly separated.

The whole cycle we just summarised provides an analytical framework to observe how each new issue is dealt with by the collective, from the initial challenge and perplexity to the moment it is streamlined through sufficient changes. But entities may also be rejected at any of the four stages, kept out of the political collective and remain outside of it, as untreated issues. We can see them, using Latour's phrase as awaiting a second ruling. Maybe they will come back, like in the case of wolves, which went extinct in France in the 1940s, but came back (from Italy) in the early 1990s and are now well established again (even though they cause quite a stir in the "collective"!). Maybe they will be gone forever, like the dodo bird. The title of the periodic meeting of francophone conservation scientists – "Waking up the dodo" – wittily reflects the status of biodiversity conservation in this (cosmo)political cycle: speaking for ecological entities that have a problem being taken into account in the collective, or are at a risk of being expelled.

9.3. ETBs in the light of "The Politics of Nature"

Politics of Nature offers a model to describe how ecological issues get to be taken into account (or not) through a process that is at the same time scientific, political, social, cultural, etc. It offers a framework and vocabulary for following in detail the multidimensional processes involved in the treatment of biodiversity issues without letting clumsier *a priori* categorisation (social versus natural, facts versus values, etc.) obliterate the finely woven processes of decision-making. As in all models of processes in stages, there are obviously overlaps and backward moves. Taking these into account, the framework and vocabulary fit very well the dynamics of how environmental issues make their way into society. The model also has a normative ambition, as reflected by the imperative formulation of Latour's principles cited above. The clumsy categorisations that ascribe science and politics to airtight spheres that would have us get a fully clear grasp on facts before discussing values, not only confuse our understanding of how decision-making processes work, but they can also encourage us to make mistakes in designing our procedures for deciding.

Letting go of simplistic separations of roles helps better analyse how the various roles of the scientist, the politician, the militant, the economist, etc. are exercised jointly at all stages in dealing with new issues. This does not mean, however, that there would be a confusion of roles, but just that they must be conceived in a more detailed and subtle way than they are in the usual view of science/politics relations. Let us now examine what light the politics of nature model can shed on the use of ETBs in managing biodiversity.

9.3.1. *Economists as accountants handing a mirror to society to examine its attachments and hierarchies*

We may start by the Latour's (2004) analysis of the role of economists in the life of the collective.

A starting point is the rejection of the usual view of the economist translating facts provided by scientists into economic language and then handing it over to politicians for making decisions based on values. This linear view does not reflect the actual dynamics of decision-making. And from a normative point of view, just as is the case for scientists, it puts the economist in a position to preempt the political debate, making covertly political decisions himself and leaving insufficient room for manoeuvre to politicians.

In the Politics of Nature model, just like the scientist, the economist has no stage that would be all his own in the decision-making process, but intervenes at all stages of the collective dealing with an issue, alongside with the scientist, the politician, etc. Of this, the detailed discussion of the roles of ESV in the next section will provide examples and an in-depth discussion. For now we would like to insist on the overall perspective that the Politics of Nature model provides to grasp the economist's role. In a nutshell, it is to keep accounts of what is internalised and what is not in the collective. The defining traits of these accounts are their systematic and explicit character. They imply models that explain how individual values add up in societal hierarchies. Feeding this template of explicited values back to the collective, the economist plays a specific and useful role – of an accountant of sorts – in the political discussions within the collective.

Latour insists much on the problems raised by positing the economist, as it were, at the wrong distance from the collective (*e.g.* from political debate). He is too close when he thinks (or other people believe) that his models describe how the collective really operates. A political collective just does not function on the basis of the models of economic theory. These are accounting models and they describe the fundamental mechanisms of the collective no more than accounts describe the functioning of an organisation or suffice to design its strategies. Much more than accounts are needed to manage an organisation! The economist is also too close when he claims to account for things and processes that are not in the reach of his measurement instrumentation. Like the accountant's, his instruments are explicit and consistent, but only within a certain domain of relevance. What is not in the reach of the accountant's measurement instruments is out of his domain. Then, the economist is too far from the collective when his accounts are not part of the discussion. In that case, there may be a lack of reflexivity of the collective on its prioritisation of people and things, for instance when no consideration is given to the costs involved in a given conservation policy or when – as we have seen in chapter one – a development project proceeds although it is completely unsound in terms of its costs and benefits, as well as loss of ecosystem services, because the promoters manage to keep the political discussion under the radar of in-depth scrutiny and evaluation.

That latter risk – a deficit in accounting – is the oft-expressed concern of many environmental economists, when they claim that we should account more explicitly for the loss of ecosystem services, or that we should be more transparent about the priorities implicit in our environmental or other policies. The former – confusing accounting models with how society really operates – is at the centre of attacks on environmental economics, as critics accuse it of reducing complex social-ecological systems to accounts where crucial facts, values and relations are lost. Latour would like to appease such concerns. For him, anyone who observes how the economists generate and treat their data and models can see plainly how huge is the disproportion between the simplicity of the assumptions used on the side of economic templates and calculus, and on the side of real life the proliferation of connections between humans (and non-humans) that makes up the fabric of environmental issues. However, he argues, this weakness turns into strength in two ways. First, hardly anyone today could misconstrue economic models with complex reality or think that they can predict the dynamics of real-life cases. The risk of conferring them an excessive status can only recede. Then, it is precisely by handing the collective a very partial, sketchy, but explicit and coherent account of its attachments and priorities that economists can play a useful role in connection with scientists, politicians, artists, moralists, lawyers, etc. In brief, economists provide templates encapsulating one summary of the attachments within the collective. The important point is that these summaries should not be understood to be infrastructures of society's operation, nor substitutes standing for the overall dynamics of the collective, nor automatic decision-making mechanisms. That would be tantamount to handing over management to accountants – which is indeed a risky move! Rather, the economists' summaries should be treated as one contribution to the deliberation of the collective, alongside with others: the work of art, the legal brief, the modelling results, the environmental ethics textbook, etc.

Latour's position may be irritating to some, either because it makes too little of the economist's role, or because he still expresses too much trust in it. Its relevance lies in the fact that after dismissing the question "Should we leave the key to the house to the economists, or should we ban them from the house?", it refocuses attention on the more relevant question: "What role exactly do economists play? And what roles do we expect them to play and how?". Seeing economists as accountants in wider systems of ecological politics and ecosystem management provides a much more precise basis for reflection on their roles, and their relations with other actors than the notion that they "contribute to decision-making". It extricates us from debates where we seem to be caught between having to discuss issues within the templates of economic theory, or rejecting it as pre-empting political decision-making. Following up on TEEB's motto: "We cannot manage what we cannot measure", it avoids misunderstandings by moving to the next question: "How do we use our measures for management? And thus whom do we need to measure what, exactly?". This opens up a wide set of questions on the contribution of accountants and accounts to management, policy and politics. "What are the rules for accounting? Where is the discussion held on the exact meaning of the accounts and resulting balances? Where are managerial decisions taken, by whom, how, how are accounts used? Do the accountants participate in the decision, or are they providing external audit-type guarantees to decision-makers?" Economics of biodiversity as an accounting issue connects directly with part of the field, like green national accounts, the balances of cost-benefits analysis, or the calculated equivalences of biodiversity banking. Beyond being immediately useful as an enlightening metaphor, it opens up en entire avenue of research that combines ecological management and policy on the one hand and the theory of accounting on the other.

As for the four-stage model of making decisions on issues combining science, politics, nature, policy, it also provides useful insights on and for the practical use of ETBs. Let us examine in that light ecosystem services valuation first, and then the various kinds of "money on the table" biodiversity management tools.

9.3.2. *Ecosystem services valuation: different roles at different stages of internalisation*

As we have seen in chapter one, the most salient results of reviews of the ESV literature focusing on the actual use of ESV are (1) the rarity of well-documented actual use for decision-making and (2) the heterogeneity of methods and claimed uses. Maybe the different stages of deliberation in the collective of the Politics of Nature model can help explain some of the problems met using ESV, and some of its heterogeneity?

A first role ESV can play is in triggering *perplexity* by alerting stakeholders to the high economic value of certain disregarded ecological entities. ESV can carry the message "Stop and think, you may have undervalued some important but not so easily computable connections." The best known example may be the paper of Constanza *et al.* (1997), calculating the total economic value of the biosphere. The controversy it has triggered reflects very well that stage of perplexity: a mixture of whistle-blowing, high level of attention, puzzlement and serious discussion while wondering what should be taken seriously.

ESV also has a rather straightforward role to play in the *consultation* phase. One of the strong points of in-depth, case-specific valuation studies is that they create inventories of interested parties, of what they have to win or lose, of the consequences of various options to members of the collective, and to the collective as a whole. They can make very explicit interests and connections that have little explicit political voice and would otherwise be more easily dismissed. This contribution directly fits the needs of consultation: identification of affected parties and of how and how much they are affected by an issue and various options for change.

In these first two stages, the role of ESV is quite straightforward. These stages correspond to the first level of the "politics of nature's" model: deliberation to decide in principle whether a given ecological issue should be internalised into the collective. These are "awareness raising" uses of ESV, such as are often put forward by the literature. The model, however, points to the vast difference between the two stages, both in terms of how ESV is used, and of relevant valuation methodology and in the processes associated with valuation. In the perplexity stage, simple (but not irrelevant!) methodologies leading to convincing orders of magnitude are relevant as well as valuations that appear as manifestos by groups or networks of experts. In the consultation stage, relevance rather lies in the systematic and meticulous aspects of field-relevant valuation, as well as in the participation of stakeholders in the valuation process through any of the many methodologies that encourage it. Both elements favour the careful inclusion of members of the collective (human or non-human) in the deliberation, in preparation of the later prioritisation phase.

The situation seems to us very different as we move from decisions of principle to the *prioritisation* stage, *i.e.* to decision-making on actual changes to accommodate the demands of an environmental issue. This is the blind spot of ESV use and, as we showed in chapter one, very little documented use for actual decision-making has been reported . One central, fundamental reason for this, in our view, is that decisions that are political (in the widest possible sense) are largely negotiated decisions. And the role of economic evaluation in negotiations is highly problematic. Negotiation by nature rests on composite and ambiguous assemblages of rationales and arguments. The part of the explicit and the implicit is never clearly established. This creates a context at the antipodes of what economic valuation is good at: univocal, systematic, explicit, coherent computation and prioritisation of values. What negotiators often need is a content-rich set of value information that they can use in a tactical manner and that leaves them room for manoeuvre. If they use valuation and priorities, these have to be set in forms that can accommodate heterogeneous and partly covert sets of concerns to the last minute of the negotiation process. In such negotiations, the most obvious uses of valuation are (a) to provide data and figures to be used piecemeal amongst other information in the course of defending positions or building compromise and (b) to put forward elements of reasoning about prioritisation (based on economic modelling of choice situations), arguments that will be extracted from the valuation's framework to support wider and more composite lines of argumentation in the negotiation. Such fragmentary use is difficult to analyse. Furthermore, due to the very nature of negotiation (ambiguity, implicit reasoning, heterogeneous rationales, poor access to observation) it is particularly difficult to document the use and impact of a given (ESV based) argument that appears in a negotiation process.

If we look at exceptions in which valuation is actually used in making hard, change-laden decisions, we find those decision-making processes where valuation has been chosen as a constitutive component of the decision-making procedure. This is the case for instance when valuation has been chosen by a court as the way to determine ecological damage compensation, as in the precedent-setting case of the Exxon Valdez oil spill. Not that there would be no negotiations in such cases, but the valuation itself becomes the *locus negociendi*, where experts and parties negotiate indirectly, valuation data, methods and result interpretation serving as media for negotiation. Another, very different example is the one described by Claude Henry (1990) of economic valuation including that of ecosystem services being used by the British parliament to support the process of deciding whether to build or not an estuary dam with large ecological impacts. In such cases, ESV is used because parties exercising effective political control over the decision-making process have made it one decisive channel in that decision-making process.

This reinforces one of our recommendations to overcome the deficit of use of ESV for the actual decision-making: it is not reasonable to expect that an unsollicited valuation should have much chance to be used by the decision-makers. As we underlined in chapter one, ESV is essentially a supply-driven field and ESV researchers may now more profitably concentrate on the formation of demand for valuation and its exact decision-making contexts.

Finally, as we turn to the institution phase, we find yet another role of ESV, with the numerous examples of using valuation to justify decisions and policies that have already been decided on. At this stage, there is an important activity of ex-post rationalisation to mend up the fractures that the tumultuous controversies of perplexity, consultation and reprioritisation is likely to have caused in the collective. Maybe the most eloquent example we have come accross is the case of a decision to remove a dam (Gowan *et al.*, 2006) in which ESV was used to convince operators to implement a policy that had been decided, but to which they had not (yet) conferred enough legitimacy to motivate them to effective action.

Between these last two stages, we find again that contexts are very different and call for different valuation methodologies and processes. In the prioritisation stage, the main criterion for use is to fit the very context-sensitive needs of one or several negotiators at some stage in the process. If one bears in mind the very volatile character of negotiation processes and the rapidly changing needs of negotiators, one starts to realise that fitness of evaluation to its context of use can then become quite elusive. A valuation framework, a data set that may be serving me at one stage can become problematic later, depending on new developments in the negotiation. In such circumstances, it may become difficult to reconcile valuation studies' time-frame and need for stability of framing and the volatility of political processes. A logical exception is those (rare) cases where valuation is chosen by those exercising power as the stable red line that will coordinate actors positions in the negotiation process. Valuation used by negotiators can be partial, patchy, as long as it provides the tactically useful information in the right form at the right time. In the institution phase, on the contrary, valuation is strong if it provides a systematic overview that helps members of the collective (at least, those who read valuations…) strengthen their sense of coherence and cohesion of the collective. Studies launched by institutions like the French National Parks service to verify, as it were, the positive economic contribution of national parks are typical of this role .

This rapid overview makes the status of ESV somewhat clearer *qua* auxiliary to decision-making. In the two decision preparation phases (perplexity and consultation) and in the consolidation phase (institution), valuation has quite different but rather straightforward roles to play. In the decision-making itself (prioritisation), its role is much more elusive and unstable, due to the composite, fragmented and volatile nature of decision-making processes that are laden with political and other interests.

From the ESV user's perspective, Latour's Politics of Nature model can help clarify, differentiate and analyse decision-making contexts, how they change as decisions proceed, and what specific needs for valuation arise. For the field as a whole, these conclusions lead to recommend further research on ESV use contexts, and suggest the two following orientations: First, prior to detailed investigation, one should distinguish clearly between the different roles played in the different stages of decision preparation or consolidation, and to differentiate expectations about ESV use accordingly. Second, it should become a priority to study the "hot" stage of decision-making – the complex, unstable and ambiguous process of political decision-making – on its own terms. The elements of exploration and rationalisation that underlie most approaches to applied valuation may be fit for the preparatory and consolidation phases, but they are not really relevant to the hot decision-making stage. For the latter, much progress could accrue from research focusing on the study of valuation use in negotiation processes and on reconnecting research on ESV use with real-life studies of political environmental decision-making.

9.3.3. *PES, buying rights and biodiversity banking*

As we now turn to "money on the table" economic tools – payments for ecosystem services, the buying of land or land-based rights, and biodiversity banking – the picture appears quite different in that they are directly at home in the prioritisation stage, when some interests in the collective are tangibly promoted, others demoted, to concretely integrate changes in balances and priorities. By nature, each of these ETB types is about transferring value and modifying the balance of interests in the collective in a tangible manner. In PES, the idea is for the payer to promote his priorities by influencing the priorities of the payee. In buying land and rights for biodiversity, conservation ownership creates new leverage that is going to change the subsequent negotiations. Biodiversity banking also implements a balance of priorities: "if biodiversity is not the priority where you build this development, then you have to make it the first priority somewhere else". Compared to offsets, biodiversity banking allows more economic, rather than administrative, fine-tuning of priorities in terms of choice of location and method of ecosystem restoration, for instance. Let us now examine in more detail how each family of tools is likely to play out in the two stages of the second level of the politics of nature model: prioritisation and institution. Not that the first two stages – perplexity and consultation – would not be interesting to consider, but we will rather let the reader transpose teachings on these two stages from ESV to the other ETBs and concentrate on those parts of the decision-making process where "money on the table" ETBs differ squarely from ESV.

Regarding PES, a fundamental observation is that the very characteristics of the rhapsodic suite of negotiations that constitutes political decision-making and that provides such a troubling context for the use of ESV creates just the right context for payments based on direct transactions between actors. Transferring money in exchange for concessions of any nature, without having to make explicit which equation this is exactly resolving, is a well- established capacity of negotiation processes. In our view, this is precisely the strong point of PES. The possibility for those who negotiate for biodiversity to make payments opens possibilities of new deals and in many cases has concrete potential for tilting the scale of agreement toward ecosystem conservation or restoration.

This characterisation of the prioritisation phase explains some of the observations on problems of use that emerge from the ESV and PES literatures.

It provides, for example, a possible explanation for the lack of use of ESV to establish levels of payments in PES schemes. PES schemes emerge from successful, often complex or sensitive negotiations; (a) they do not need valuation (one can obviously agree to a payment in a negotiation without basing it on an evaluation) and (b) political negotiations, as we discussed above, offer a difficult context for the use of ESV.

It can also account for the extreme heterogeneity of PES as a family of tools, as reflected by the – sometimes highly puzzling – debates on the definition of PES we presented in chapter two. Rather than reflecting a theoretically unified logic, PES are responses to negotiation situations about biodiversity. The great heterogeneity in PES schemes reflects the heterogeneity of such negotiation situations. There are indeed many very different sorts of situations and negotiations where a payment can make a difference. The logical consequence is that to analyse PES use, it is more relevant to focus analysis on the entire dynamic of biodiversity deals, than to focus on the payments *per se* as if they were by themselves the consistent organising force of the action.

Other problematic aspects of PES come into focus as we envisage the transition from the prioritisation phase to the institution phase. This transition echoes the crucial issue of durability of PES schemes. The strength of PES as an instrument that can work comparatively rapidly and flexibly through negotiated deals becomes a liability in the longer term. Deals can be easily changed, if they are not institutionalised; voluntary payments are difficult to extend over time, if they are not integrated in some way in the basic rules of the economic game. In terms of the politics of nature model, since new issues appear on the agenda all the time, we should not (and usually cannot) rely on permanent negotiation to manage each and every long-standing issue: we have to consolidate the negotiation result, move it out of what is being negotiated and move on to use our (limited) negotiating capabilities on some other problem. In the PES literature, a first treatment of this issue is through the distinction between those PES schemes that are likely to require indefinite payment, and those which are not, because the payments are used for transitioning from one set of practices to another, that will be more sustainable and, once established, will not require indefinite payments. In such cases, once the transition is over, the new perimeter and priorities in the collective are streamlined: it can function sustainably and maybe turn its attention to new issues. A second, more implicit treatment of this issue is evident in the difference between the archetypal form of PES (just a business deal) and more institutionalised forms such as the agro-environmental payments of the European agricultural policy. These are payments for ecosystem services, but they have become an integral part of the basic rules of the game of farming as established for extended periods by the CAP and they are renegotiated with all the rest of the rules each time the CAP is revised.

The same passage from prioritisation to institution accounts quite nicely for the paradoxes we had noted about *buying land or rights* for biodiversity, a tool that requires high levels of goodwill to be set up, and once there, is highly constraining. The process leading to land or rights purchase has all the characteristics of the prioritisation phase. It involves multiple negotiations. As we noted in chapter three, although the buying could be seen as a simple market transaction, in most field situations it requires many other associated negotiations (for political support, regulatory adaptations, fiscal incentives, etc.). Many aspects of the collective will be set in motion simultaneously as the acquisition of rights for conservation brings lasting changes in the priority structure of land management. Once the rights or the land are bought, that change is embedded in the institution of private property. In those countries where this is a robust institution, the context changes dramatically: the property of the acquired area is no longer a matter for discussion, but part of the basic new rules of the game. The actors' attention turns to many other topics for negotiation: multiple use, land management, ecosystem management, etc. In countries with more fragile institutions and problematic property regimes, this closure can become difficult or impossible. This then becomes a major consideration in the use of tools based on the acquisition of land or rights, so that based on a detailed context analysis, one should prefer other types of tools, or concentrate implementation efforts on the institutionalisation phase.

As for *biodiversity banking*, we showed in chapter four to what extent it relies on very robust institutional commitments to balance priorities regarding development and conservation (such as the "no net loss" principle or other commitments to a firm target). For the tool to be used effectively, such commitments have to be built over time through a demanding political process. In 1994, as we presented the results of a research evaluating French wetland policies (Mermet, 1996) to a panel of experts and officials, the most senior official in the room turned and cried out to no-one in particular: "but we aren't going to keep them all [French wetlands], are we?". That is perplexity. Moving from that point to a robust collective commitment to stabilise biodiversity and ecosystem services implies years, maybe decades of more perplexity, consultation and prioritisation. Only then will the scene be ready to create the institutional basis for an effective market for biodiversity banking.

Conclusion

Latour's Politics of Nature propose a model of how ecological issues get integrated into the life of political communities through composite political processes. The parallel between this model and the internalisation of externalities that is at the root of environmental economics is quite striking. The Politics of Nature model can be seen as an extension of the idea of internalisation of externalities beyond economic internalisation to include all the technical, social, political dimensions of internalisation. It is very consistent with the observation we repeatedly made on the use of various ETBs in the first four chapters: far from working on their own, if they are working at all, they are setting in motion simultaneously all the processes (social, political, managerial, cultural, etc.) of solving collective problems. The particular contribution of the Politics of Nature model is to provide a structured framework and enlightening concepts to follow these intricate processes, without imposing artificial separations that would lock economics, science, technology, politics, etc. in separate spheres.

Callon's translation model makes a similar contribution, but seen from a different perspective. It helps us get as close as possible to the point of view of the innovator as he busies himself in the attempt to make way for some new entity into the social fabric. This is the point of view of the tool user rather than of an observer overlooking the entire construction site as works proceed to deal with an ecological issue.

Overall, the two models share the same theoretical foundations of actor-network theory. It is important to note that these are essentially approaches that provide concepts and models for more refined observation of processes of innovation and decision-making. Both authors are very clear that the crux is to observe in the field how the members of the collective behave and relate to each other, what meaning they give to events, to their own actions and to what others do. The researcher has no overbearing template to reveal the meaning of such processes; the best he can do is to recount as precisely as possible how the actors themselves give meaning to situations and how they transform them through the networks of connections they make or break. By contrast with common-pool resources theory for instance, such approaches have little claim to providing recommendations on how to manage eco-systems. If they venture into normative indications, these are of a very general nature, such as reflected by Latour's four imperatives quoted above. These he presents as a new constitution – ground rules to help us get away from the old regime of separate spheres for science and for technology, for science and politics, for nature and society – we might add, for economics and politics – and move to a new regime in which ecological issues are managed alongside all the other business of society. The general thrust of these principles is to be careful to keep an open mind on what may be the fact and to keep decision-making processes open to involvement of parties in the deliberations leading to decision. In this chapter we have tried to show, however, that beyond their relevance for observers, Callon's and Latour's conceptual models are also useful for guiding ETB users in action, as they work to make their tool an "obligatory passage" fruitful for all or as they puzzle on the changing contexts to which their tools must be relevant.

We would like to underline also that, although actor-network theory puts a strong emphasis on process, it should not lead to adopting a purely or a mostly procedural perspective. To create new connections, one has to undergo intense interactions, but there also has to be a substantive fit between the entities that are connected. For instance if a certain farming system cannot (from a technical or economic point of view) deliver the biodiversity benefits that are expected, then no amount of process (dialogue, mediation, participation) will be able to overcome that. Substance and process are equally important to consider as we embark on creating new connections to sustain social and ecological systems.

Finally, it is worth returning to the fact that both Callon's and Latour's approaches developed from concerns about innovation and sociology of science. Not only do they encourage and help us to link economics and politics, but also in turn to link both with science and technology. The connection with science is quite present already in the biodiversity and ecosystem services agenda. Experts in ecology, in environmental policy and in environmental or ecological economics have a (comparatively) long history of collaboration. The technological aspect, however, is much less salient although its importance is obvious. Choices of farming techniques, of forestry methods and of hydraulic development are decisive for the dynamics of biodiversity. Both models draw our attention to the fact that our management of biodiversity is the joint product of our science, our politics, our economics and our technologies. Technological changes are a constitutive part of most sets of tools in conservation situations. Their synergy (or antagonism) with economic tools should rise on the agenda of ETBs and actor-network theory based approaches are very relevant for that task too.

10. Strategic environmental management analysis: the antagonistic component of acting for biodiversity

To open this last chapter of the book, let the reader consider some biodiversity cases he or she is familiar with. Whether they are about deforestation, marine mammals, the conservation of traditional agro-ecological landscapes, etc., it is very likely that the struggle of a comparatively small group of actors committed to biodiversity has played a central role in the process that decided such cases. Often, the struggle is rather frontal, against powerful actors – loggers, intensive farming organisations, dam-builders, poaching networks, etc. – whose projects threaten some element of biodiversity. Sometimes it is more diffuse, for instance when actors motivated by biodiversity struggle over the years to revive extensive farming or forestry systems that have positive impacts on biodiversity but are eroding under various societal and economic pressures. Strategic environmental management analysis (SEMA) considers the struggle by these biodiversity-motivated actors to be the crux of biodiversity management.

SEMA originates from our own research on the foundations of environmental management (Mermet, 1992; Mermet *et al.*, 2005; Gaudefroy de Mombynes, 2007). It offers a framework to analyse environmental situations in a way that focuses on the question: "In this situation, what strategic diagnostic could help an actor who is ready to struggle in favour of the environment to make a difference by a relevant choice of strategy?" A major characteristic of SEMA is that it breaks away from analysis and discourse founded on collaboration (Mermet, 2011). Here, the "we" in "we should act to conserve or restore biodiversity" means this first of all: those of us who want to act for biodiversity should be ready to act strategically so as to bring those of us who don't (or don't care) to change their behaviour, their projects or their decisions.

This asymmetry is important to analyse the use of ETBs. In chapter six, we already discussed in detail the importance of establishing who exactly is using the tool and on whom – or in other words, of who uses the tool to do what to whom. In this chapter, we will examine how the SEMA framework can be used to analyse strategically the context in which an economic tool is used and the strategy of actors acting in favour of the environment, a strategy of which any ETB is just one element, however important it may be in some cases. We will first present the SEMA framework and its major concepts that may be of use when analysing ETB use situations. We will then turn to several possible uses of the framework for such analysis: (1) general strategic diagnostic of situations where ETBs are used or considered, (2) focusing on critical economic analysis of activities that are detrimental to biodiversity, (3) identifying situations where focusing too much on the collaborative dimension of processes for change may lead to missing decisive elements of context or to choosing ineffective action strategies and (4) understanding the need for and the dynamics at play in the construction of a strong, specialised, biodiversity management sector.

10.1. An introduction to Strategic Environmental Management Analysis [33]

The origin of the SEMA goes back to the early eighties (Mermet, 1992; Mermet, 2007*b*). The framework and its main concepts were proposed (1) based on the observation that the most instrumental actors in most biodiversity cases were rather specialised groups or networks struggling in favour of conservation, irrespective of their status (NGOs, civil servants in environmental ministries or agencies, professionals embracing an environmental cause, etc.) and (2) as a reaction to the then quickly developing collaborative approaches (environmental mediation, *gestion patrimoniale*, etc.). These two points seem to us just as relevant today as they were then. The first – the relevance of actors specialised in action in favour of biodiversity – has always been, and still is the object of debate. It is criticised by those who think integrated strategies are always preferable, or those who advocate that biodiversity should best be taken in charge by the very actors who have most impact on it. It is also an object of struggle, for instance as administrative reforms merge comparatively autonomous environmental ministries with ministries having in charge the agricultural or (as has been the case in France since 2007) the public works sector. It remains, nevertheless, that in the field, specialised action in favour of biodiversity is as salient as ever, and

[33] This section is largely based on Mermet 2011.

that strategies that would be integrated from scratch (*i.e.* that would rely entirely on biodiversity being taken in charge by non biodiversity-focused operators) have not proven their effectiveness. As for the second point, over the last thirty years collaborative approaches to environmental problems have risen from being innovative experiments and bold ideas of the 1980s to being at present hegemonic in discourse and omnipresent in practice. The risks from their blind spots have become proportionally more worrying.

10.1.1. *A framework to address the blind spots of collaborative approaches*

Let us consider three of these blind spots.

The first is the fuzzy view of agency associated with collaborative approaches. When defending the need for a "diagnostician [able] to match governance arrangements to specific problems embedded in social-ecological contexts" (Ostrom, 2007), or for someone in a position to "get the incentives right" so that people can "be induced to make production and consumption choices that are relatively less stressful to the environment" (Clark *et al.,* 2005), where exactly does one see this character standing in the system, and from where does he derive the power to change governance arrangements or what people do? This is the issue of agency we treated at length in chapter 6 – an issue to which our preference for the strategic perspective makes us particularly sensitive.

A second blindspot is the irenic view of strategy, *i.e.* the disowning of environmental conflict by presenting it, for instance, as belonging to a former area when inadequate "traditional competitive framing" staged "a contest between environmental protection and human development" (Clark *et al.,* 2005) or when "conventional resource management [was] pitting stakeholder groups against one another" (Armitage *et al.,* 2009). Seen from the field, it seems to us that, whereas examples of collaborative action in favour of biodiversity have become much more numerous, this does not mean that there are less biodiversity related struggles nor that they have become less decisive.

A third blind spot results from the fact that any collaborative approach tends towards some form of closure of the social-ecological system. For the actors "to sit around the table" and manage together, there has to be a limited list of actors and of scope. This is a feature, for instance, of common-pool resource theory, where limits in the number or variety of actors, as well as clearly identified boundaries to the resource system are identified as some of the main factors favouring sound commons management. Many biodiversity issues, however, completely overflow such treatment, as chains of action and management extend from local to global and sector to sector, as ecological processes cross boundaries. The ideal model of a community jointly managing its resources is still relevant for many issues of local renewable resource management, or for some larger-scale resource management situation involving a limited number or variety of players, but it would be highly problematic as a general model.

As we shall see in a further section, identifying such blind spots is of direct importance for ETB uses that rely on assuming collaborative contexts or actions. For now, it helps introduce three guiding aims of SEMA: to guide analysis of strategic agency in favour of biodiversity, to provide appropriate focus on the adversarial dimension of biodiversity management and to allow strategic analysis for action in contexts without clear boundaries of their own and the complexity of which overwhelms the possibility of a diagnostic that may be shared by all actors.

To reach these aims, the SEMA framework posits five framing principles, and proposes a set of concepts to provide them with a lexicon.

10.1.2. *1st principle: base analysis on one clearly defined biodiversity concern*

When taking up a case to be analysed, SEMA proposes to found its analysis on one, clearly defined, environmental concern. For instance, if an environmental actor undertakes to conserve a forested watershed to ensure ecosystem services in terms of water quality, the entire analysis of the action system will be constructed based on that reference concern. This offers much flexibility in the analysis of multi-scale and fuzzy perimeter socio-ecological systems because one has to explore only those chains of causality and agency that are relevant to the one chosen reference concern. Because one has a very clear criterion of what is relevant in the analysis, one can go far and deep in the analysis of these chains of causality and agency. The cost associated with this flexibility and depth is a limit in breadth: the analysis is indexed on one concern and perspective only. This would be a serious problem in cases where there existed only one diagnostic and analysis, and that was based only on a biodiversity concern. But in most cases, there are plenty of other diagnostics and analyses that are based on other-than-biodiversity concerns and commissioned for example by planning authorities, by the farming or forestry sectors, by economic development agencies, etc. In a given case, SEMA-based analysis has no claim to replace other analyses based on competing concerns, but only to make sure that biodiversity concerns receive the best possible diagnostic on the causes of given biodiversity problems and on what is required and possible to solve them. Other concerns and perspectives can commission analyses of their own, and debate will occur as a result, not as a pre-condition of such plural analyses [34].

10.1.3. 2nd *principle: un-bundle the ambiguities of "management"*

Based on the clear definition of a reference concern in terms of biodiversity it becomes possible to un-bundle the ambiguous concept of management as we use it for instance when writing about ecosystem management. Our habitual use of management as a concept tends to merge agreement on expected performance, joint accountability and coordinated action. This is acceptable in situations where there is, in actual practice, a fair degree of convergence on goals and coordination of actions, and of alignment between commitments on aims, actors' strategies and actual practice. Most real-life biodiversity management situations are just too far from such convergence and alignment for such an ambiguous concept of management to be useful. To account for it, we propose a dual, dialectic concept of management that defines and treats two dimensions of management separately. (a) Actual, *i.e. de facto*, management of an ecosystem is the whole set of anthropic actions that, whether the actors realise it or not, whether it was their intent or not, have a decisive influence on the ecological condition of the system (more precisely, of those aspects of that condition that constitute the expected environmental performance – more on this below). Its analysis includes identification of mechanisms by which these influences are exercised and of the places where the actions with the most significant impacts are decided. (b) Intentional management – which could also be called interventional management – is the set of actions that have as their main and explicit aim to reach expected environmental performance.

Consider for instance the management of river water quality. If one starts from the principle that joint accountability exists for water quality in a river, then a factory discharging pollution, a dam intercepting part of the low-ebb river flow, a sewage purification system, a farming policy subsidising irrigation systems that pump water from the river and a series of demonstrations against ongoing water polluting activities, are all examples of management actions that are decisive for a river's condition; thus they should all be held accountable and considered part of actual

[34] Discussion of this perspective on pluralism lies beyond the scope of this book. (see for instance Mermet, 2011).

management. In the above list, only the construction of sewage purification systems and the demonstration against pollution could be considered as int(erv)entional management.

These definitions may be puzzling to those who see ecosystem management as the set of institutions and policies that have been agreed upon to attempt to tackle the environmental issues faced by a particular ecosystem: international environmental regimes, integrated management institutions (for a watershed, coastal area, etc.) and the like. But much that is decisive for the ecosystem – and thus for management accountability – occurs outside of such instituted management systems, through cross-scale linkages that may be ecological, social, political, economic, etc. (Armitage *et al.,* 2009). In addition, many aspects of action to change the course of ecosystem degradation are excluded from such a management view, *e.g.* the actions of environmental activists, which many case studies show are instrumental in the inception of a collective capacity to steer away from unsustainable courses. SEMA proposes to set a wider framework, encompassing the entire dialectic between actual, *de facto* management and strategic interventions for change (int(erv)entional management). This is the momentous process through which the future of an ecosystem is played out in the long run. In a given case, current institutionalised management arrangements, as they have evolved over time from that very dialectic, form a part of that picture; a part that varies in importance and may be incomplete or sometimes deceptive, depending on how close the field situation is to a hypothetical unity of expectations, accountability and action. At any rate, centring analysis on the current set of institutionalised management arrangements provides no guarantee of a sound diagnostic investigation into the management of an ecosystem or environmental problem. Furthermore – and this will have important consequences for the use of ETBs – the problem is only very rarely a lack of institutions and organisational presence, but rather, of too many (partly contradictory) institutions and (competing organisations), so that there is hardly any room for introducing new rules and organised interest without challenging the existing ones.

10.1.4. 3rd *principle: focus analysis for action on actors who intervene in favour of biodiversity*

The primary need for management thus is not for all to agree on how to remedy a vacuum in rules and organisations, but for someone to intervene on existing arrangements with such motivation and adequate strategy as to be able to obtain changes in the established interests and strategies of others. This calls for an analytical focus on the actor(s) who focus on defending the specific environmental concern that founds the analysis. We already mentioned earlier how this focus reflects the dynamics of most field cases of biodiversity issues. In analytical terms, it amounts to introducing a differentiation amongst actors that echoes the unbundling of management we just discussed. Management of ecosystems and their services, of biodiversity, is then seen as playing out between (a) an environmental actor who acts in favour of conservation (it can be a lone militant, but in contemporary situations it is more typically a complex network or coalition of forces (Taravella, 2008), (b) sector-based actors (foresters and the wood industry, farmers and the agricultural complex, mining companies, etc.) whose activities are the main determinant of the actual condition and transformations of biodiversity and (c) regulating actors (a government or governor, a mediator, etc.) who exercise a mandate to try to balance diverging interests of actors in society. These various groupings are relevant only in view of one clearly defined reference concern – for instance hydroelectricity producers can align themselves as environmental actors relative to climate change as an environmental concern, whereas they are clearly on the side of the sector-based problem relative to aquatic biodiversity concerns (Gaudefroy de Mombynes, 2007).

These three types of actors have a completely different relationship to action in favour of biodiversity and thus to the use of tools like ETBs for biodiversity management.

In a SEMA perspective, regulating actors play an important but not really decisive role, in that they essentially can institutionalise a given balance in power relations, they may influence it somewhat, but the balance (*i.e.* the level of environmental protection that will be instituted) results mostly from the struggle between environmental actors and sector-based actors. Furthermore, we feel that regulating actors is the centre of attention in most approaches to environmental policy and management, so there is a need to rebalance the focus on concentrating rather more on environmental and sector-based actors.

10.1.5. *4th principle: focus analysis of situations on activity sectors*

So after the focus on the environmental actor, the fourth framing principle of SEMA is to focus also on activity sectors, as they are the fundamental organisational structure of activities that impact biodiversity. In the management of biodiversity, changing behaviour that harms biodiversity is a central concern. One common example is the changing of farming practices that generate water pollution or losses in the diversity of vegetation. Practical experience and field studies soon show, however, that the practices of one farmer, as well as his production system at the farm level, are very difficult to change on an individual basis. His choices are set in a wider context that includes the industry's technical support chain, trading organisation and market conditions, the training and culture of farming sector organisations and unions, all of which are set within the framework of rules and subsidies enforced by the agricultural administration. The farming sector thus functions as a large, partly informal, but functionally coordinated organisation of collective action, in which technical, economic, educational, legal and administrative components share essential concerns and actively coordinate their actions and strategies. This type of organisation extends from the farm to the global level (Food and Agriculture Organization, World Trade Organization), through all intermediary levels with local, regional, national and supra-national (*e.g.* EU) farming policies. It seems hard to overestimate the strategic importance of the organisational strength of such sectors as a major force in the biodiversity strategic force field.

In the field of biodiversity, the farming sector is not only the negative force behind the catastrophic impact the industrialisation of agriculture has had, and is having on ecosystems. In some other cases, it can be a crucial part of the solution, for instance when good ecosystem management relies on the continuation and adaptation of traditional farming practices, or on innovative production systems that provide ecological benefits, or when production techniques are part of an ecosystem restoration strategy. As biodiversity conservation also invests more and more in such partnerships with segments of the farming sector, sound strategic analysis entails the need to concentrate on the dynamics – the economics, the politics, the social and technical issues – of the farming sector.

Of course, farming was just an example here: the importance of sector-based organisation is similar in most fields that are at the heart of environmental issues: forestry, energy, transportation, building, etc. Acting to solve an environmental problem means, for the environmental strategic actor, to undertaking to effect organisational change (halting damaging practice, promoting practice that is favourable to biodiversity) in one or several sectors. Organisational and strategic links within each activity sector are a major structuring factor in the strategic force field of environmental problems. It is essential to analyse these links carefully, in addition to the now traditional consideration of local community dynamics, of national policy making or of global regime negotiations. In our view, it is surprising that "horizontal" dynamics of local community, national policy, etc. receive almost all the attention in contemporary publications that bear on the theory of biodiversity governance, whereas sector-based dynamics receive so little, although, as soon as we factor power and concrete organisation into our analysis, they are decisive.

10.1.6. 5th principle: restore its full scope to strategy by including adversarial relations

A fifth and last framing principle of SEMA is to restore strategy to its full dimension. This is needed because although the concept is used extensively today about biodiversity policies and action plans, it is used in a watered-down manner. Consider the following definition of strategy by Mintzberg (in Mintzberg *et al.*, 1995) as "the pattern or plan that integrates an organization's major goals, policies and action sequences into a cohesive whole. A well formulated strategy helps to marshal and allocate an organization's resources into a unique and viable posture based on its relative internal competencies and shortcomings, anticipated changes in the environment and contingent moves by intelligent opponents." Mintzberg insists on the multi-faceted nature of strategy as a concept and an activity (see also Mintzberg *et al.*, 2005), which he summarises through the formula "strategy as plan, ploy, pattern, position and perspective".

And consider now IUCN's 1980 "World Conservation Strategy" (IUCN, 1980) and the dozens (hundreds or thousands may be closer to the mark) of biodiversity, conservation or restoration strategies that have become a must in biodiversity action and policy today at all scales from local to global. In such documents, you do find perspective, position, pattern and plan. But the adversarial dimension of strategy tends to be attenuated, usually to the point of vanishing almost completely. In such "strategies", it is hard to find any mention of "intelligent opponents", that is, of organised actors and actions that deliberately develop resource exploitation strategies that damage ecosystems and biodiversity. "Strategies" often seem to be opposing only anonymous human shortages: lack of awareness, ignorance, insufficient coordination, etc. Inasmuch as it privileges collaborative perspectives, much of the academic literature also currently tends to underplay that dimension. When promoting integrative perspectives, it tends to posit the manager and researcher as facilitators, and often sees sustainable development as a collective participatory planning problem, that is, not the strategic problem of some actors confronting others, but of all actors jointly confronting a shared problem.

Having to deal with intelligent opponents, however, is not an optional but a fundamental dimension of strategy. Practitioners of biodiversity management in the field experience quite intensely intelligent resistance to environmentally-motivated changes as an integral part of the cause they act for and of their job description. It adds a whole new dimension on top of (or at the heart of) the complexities of collaborative environmental planning. In our view, it is essential that analysis of biodiversity issues include explicit, systematic treatment of that dimension of strategic action. The other framing principles of SEMA – the selection of a clear but partial reference norm on which to found the analysis, its dual concept of management, its clear focus on the environmental actor and then on sector-based actors – are all designed to set the stage for an analysis that is truly strategic, *i.e.* that encompasses the need to confront intelligent opponents as "we" strive to conserve or restore biodiversity.

10.2. Setting ETB use in the context of strategic action in the face of resistance to change

As the other frameworks presented in the previous chapters do, each in a different way, the SEMA framework organises questions, helps to focus on a coherent and limited set of issues and assists with (but does not replace) the choice of appropriate analytical tools and investigation methods. As we now focus again on ETBs, the main scope of SEMA is to set the use of ETBs in the context of the strategic action of actors who strive for change in favour of biodiversity, who face the opposition of some other actors who resist such change, with some level of regulating being provided by regulating actors who can provide some degree of mediation and institutionalisation but ought not to be considered as the fulcrum of action. What light does this perspective shed on the use of ETBs?

10.2.1. *Ecosystem services valuation: a compass... for advocates*

In our review of ESV in chapter one, it was quite evident that valuation has no effect by itself, as if the availability of information on revealed preferences could directly impact decision-making. If and when ESV is useful, it has to be taken up by actors and used in the dynamics of the decision-making process. The most immediate contribution of SEMA here is to draw our attention to the primary differentiation of roles between actors in dealing with biodiversity issues, between the environmental actor (whose action is in favour of the biodiversity item at stake), sector-based actors (whose actions impact that biodiversity item) and regulating actors (who intervene for co-existence and some sort of balance between opposed interests). For these three types of actors, the use of valuation does not have the same meaning; it is embedded in deeply different strategic perspectives and courses of action.

To illustrate, let us go back to the example of ESV use in the implementation of the European Water Framework Directive in France (chapter one). In the controversy between the Ministry of Environment, which pressed for more sanitation works to catch up with France's lagging results in terms of water quality, and the Ministry of Budget, which expressed concern about the increase in taxes and charges that effort would cause, each Watershed Agency (*Agence de Bassin*) conducted its own economic valuation of the benefits provided by the sanitation program. But they refused to present such valuations in terms of cost-benefit analysis (CBA) – *i.e.* they refused to present ESV as a tool for arbitration, arguing that the investment was required to reach a regulatory target, and thus, the balance between cost and benefits was not relevant to decision-making. For them the rationale behind the studies on benefits to be gained from cleaner water bodies was to be useful as justification of the regulation, and support for its implementation. This use for advocacy influenced the choice of a methodology based on case studies, that they thought best suited to demonstrate such benefits. The Ministry of Budget, on the contrary, demanded CBAs that would weigh costs and benefits on the scale of a whole watershed. This use corresponds to the "injection of rationality", a completely different sort of use as we discussed in chapter one, calling for quite different methodologies. The punchline of the case was the fact that the industry representatives did not use these results to justify voting against the expense, but voted in favour and used the valuation to show how large the sacrifice was that they were willing to make as a concession to the demands of the environmental sector. This case illustrates quite well how valuation is best understood in terms of providing argumentative ammunition to be used strategically in debates, confrontations and negotiations about biodiversity. Usefulness is essentially a matter of providing the right line of reasoning for a given actor in a given strategic situation. The needs of environmental advocates (here, the watershed agencies), of sector-based actors, and of others who intervene in the decision-making are different from one another, and they are different from one case to another, because the strategic configuration can be very different.

Overall, situations where valuations are used as advocacy (or justification which is, in effect ex-post advocacy) clearly dominate the scene of ESV use. The most relevant point for practice is to call attention to the diversity of advocacy situations. If ESVs are to be fit for use, they require a clear, detailed, diagnostic of by whom exactly, for what and in what context they are to be used. In a case like the one we just summarised, there are already several differentiated uses requiring different valuations. As one considers the variety of biodiversity management cases, the diversity of situations becomes great indeed. To scope this variety, to posit better diagnostics is one of the great challenges that awaits the ESV field. SEMA structures such diagnostics through three questions: (1) by who exactly, (2) for what exactly (3) and in what context of (partly adversarial) interaction is a given ESV to be used.

In the context of ESV use for advocacy, the fit of valuation to use means a fit of the framing questions of the evaluation, and of the methodologies to the advocacy context in that particular case. If you want to prove that conservation can deliver high benefits to local populations, it is perfectly fair to look for case studies that will support that point. If you counter-argue that they may also be a false hope for local populations, it is equally fair to elect cases that will make that point. Advocacy being the main use of ESV brings with it a certain relativity of points of view. This may be quite troubling for some experts, who would like to see valuation precisely as a way to step out of the exchange of partisan points of view and into objective balancing of interests.

The point, however, was made in a striking way in 1980 by a pioneering study by a French management research team on the actual use for decision making of economic studies of transport systems (GRETU, 1980). They organised a thorough system of anonymous interviews and panels through which all actors involved in major decisions about transport systems could testify without institutional pressure about how decisions were really made and studies utilised. The team coined the phrase "advocacy-study" to underline the fact that – much to the consternation of French engineers and economists of the early 1980s – in real decision-making, economic studies were not used as scales for arbitrage, but as resources for advocacy. Although it sounds less provocative today than it may have then, such a view of valuation may nevertheless remain problematic in the eyes of those who still view ESV as bringing a measure of objectivity and external appraisal to the struggle of actors, and useful precisely in the measure that it does so.

In our view, it should not be. As we made it clear already in chapters 1 and 6, these are perfectly legitimate aims for the use of ESV, and ESV can serve such purposes. Our point here is that these aims also have to be backed with good strategy if they are to succeed in practice. Even the roles of valuation for rational and objective economic critique, or of arbitrage for the common good, that can indeed be supported by ESV, have to be embodied in some tangible (managerial, political) strategic agency. They have to be borne out by concrete, strategic actors who will have to participate in, and withstand, the political game. As an example, this is precisely the position of the Ministry of Budget in our previous water-related case. With its concern to balance costs and benefits, that Ministry is not above the fray, but still one political actor amongst several. The fact that it adopts a point of view that encompasses the opposite positions of environmental actors and of economic interests does not result automatically in it having the power to arbitrate. In this example, the position of the ministry of budget did not prevail and the environmental sector carried the day.

In chapter one, we insisted (1) that the opposition between valuation as an objective revelation of preferences and valuation as a construction of preferences should not be overplayed and (2) that the external critique of decisions from the economist's point of view is indeed one useful possible role of ESV. What strategic analysis of the actors' relations add here is focusing attention on what it concretely entails, in the confrontation of actors, to be one actor who will try to push data on preferences to be taken into account, or who will enter the political debate to criticise this or that proposed decision or policy, based on his valuation studies. As we noted the rarity of actual use of ESV for arbitrage and decision-making, we noted a series of exceptions: the use of ESV commissioned by some courts to determine damage, or the use of ESV as the rationale and procedure for joint decision-making processes (Henry, 1984). In each such example, ESV is used because there is a real user with the right position in the decision-making process to use it, because he has an interest in using it, because he has chosen to do so, and has found the right experts who will provide studies that will be fit for the context of use in terms of content and of the precise set of power relations at work.

To sum up in view of the SEMA framework, ESV is mostly used by environmental actors in advocacy in favour of biodiversity conservation. It is sometimes used by regulating actors to help them prepare or sustain an arbitrage. And it is sometimes used by sector-based actors, or by actors supporting them, to plead for restraint in expenses in favour of biodiversity, as in the example of the Ministry of Budget in the example above, or to demonstrate the value of the efforts they make in favour of biodiversity. In all cases, use is a function of a real user and of valuation that is fit to the strategic challenge this particular user has to face. Considering the diversity of users and of strategic situations of use, useful ESV relies on a combination of a sound diagnostic of the strategic situation and intention of use, and of methodological competence on the part of the expert.

Does this sound obvious? Maybe it does to a point, if considered from the point of view of the practitioner, *e.g.* of the economist who spends his time in the field, doing valuations for clients or in support of local actors. But it does not if set side by side with academic publications on ESV or with the abundant expert-committees literature. In that literature, the fact that ESV is used as a source of information (rather than a method for making decisions) is now well acknowledged (see TEEB diagram). But the fact that in political decision-making information is used essentially as a strategic resource by struggling actors is considerably downplayed. When it does care about ESV use, either the literature focuses on collaborative uses – but in this case, from a SEMA perspective we would raise the question: who has acquired the power to launch a collaborative process, and how? – or it treats the strategic dimension mostly in an implicit way. A good example of the latter is provided by TEEB: by devoting volumes to how various concrete actors (businesses, local authorities, governments) can use ESV more than they now do, the study shows acute awareness that the main factor for use is having real-life users. But the strategic and adversarial aspects of such use remains implicit. It is common knowledge that a local authority trying to curb unsustainable use of a forested watershed will have to struggle against resistance to putting in place better management of the ecosystem. That dimension, however, is not explicitly taken into account in the ways ESV use is envisaged in publications. There is indeed some strategic wisdom in that omission, based on the belief that being explicit about power relations just makes confrontation more acute, while recalling potential gains from collaboration hopefully fosters collaboration. But this position sheds light on only one side of the coin, and has its limits (Mermet, 2011). To give just one example, it may provide cover for manipulative and damaging power play by some local actors who put up collaborative participation processes only as a front to preserve business as usual, to the detriment of better management (Mermet, 2001). In many cases, it becomes necessary to illuminate the other side of the coin: the intense power interactions which are an essential part of how we actually manage biodiversity, and thus of how we actually use ESV.

10.2.2. *Critical scrutiny of the economics of projects and development that damage ecosystem services is at least as decisive as ESV*

As one posits ESV as essentially used in various roles of strategic argumentation in favour or against a certain decision, a second look is needed at how it fits in the wider picture of economic arguments used in decision-making. Indeed, all economic valuation studies are used as advocacy, as in the example of transport studies discussed above.

Decisions about biodiversity often pitch in struggle or negotiation environmental actors and sector-based actors, the latter defending projects or development programs and strategies that may hurt biodiversity, but that they claim are essential for economic development. Asserting the economic value of the ecosystem services that may be damaged is just one side of the coin in such a confrontation. The other side is the critical analysis by economists sharing biodiversity concerns of development projects, programs and strategies.

There is a major role here for economists, who should not confine themselves to assigning monetary values to ecological services, but should partake in the wider economic debate that is part of most public decision-making. In this respect, just as

Claude Henry had in the early 1980s, we have repeatedly observed and experienced cases in which the economist played a decisive role not though ESV, but by showing that specific projects, programs or policies, with major negative ecological impacts were not justified economically, not just on the basis of biodiversity issues, but also in the very terms of the projects, programs or policies themselves. If ESV gets to be used, it can only be in the framework of the wider study and discussion of the economics of decisions affecting biodiversity. This includes the economics of farming, of forestry, of water management, of transportation, of urban development, since the "actual management" (in SEMA terms) of biodiversity is essentially driven by such sectors of activity.

However, as soon as one realises that economic argumentation cannot rely solely on ESV, but has to combine it with the economic valuation of development projects and programs, one has to question the wisdom of assigning ESV to a largely autonomous field of study, as it is now, almost confined away from where the main debate takes place: about the economic wisdom of alternative development paths (in forestry, agriculture, transport, etc.) that imply different impacts on biodiversity, or different potential for restoration and better management in the future. But once one realises the desirability of widening the scope beyond the sole scene of ESV, one also realises why doing so is difficult. In the economics of climate change, for instance, the economics of impacts of climate change on ecosystems have to be linked with the economics of emissions, in order to demonstrate where policies for the atmosphere are also questionable on economic terms. Since the economics of the energy sector and of growth overall are central in climate change issues, and since they are both well-developed fields of study in economics, this is challenging, but tractable. However, as we turn from climate change to biodiversity, pressures are associated with much more diverse fields of activity, and the impacts are connected with the economy in a much more heterogeneous way. As a result, the challenge may look much more daunting, envisaged overall.

If, however, we take seriously the decisive importance for the actual management of biodiversity of the dynamics of activity sectors, there is no way around this obligation. When focused on specific biodiversity issues, criticising potentially harmful sector-based activities is necessary; comparing various alternatives within sector-based activities is essential. Advances in taking into account ecosystem services will not be able to rely only on the economic consideration of ecosystem services themselves, but rather on a combination of an economic critique of unsustainable policies with the valuing of positive services rendered by ecosystems. Assuming that ESV provides information that can then be aggregated with economic valuations associated with sector-based economic activities amounts to assuming that these evaluations are available and that they are neutral in terms of comparing alternatives. But they are often unavailable, and when they are, they have been designed as advocacy for this or that sector-based activity. Any balance in economic valuation, especially a balance including ecosystem services, will have to be reached through confrontation between the advocates of the various sector-based alternatives and advocates of the biodiversity issues at stake. At the macro-economic level, economic policies harming the environment and the "brown subsidies [35]" associated with them are at the core of biodiversity issues and orders of magnitude more decisive than the marginal correction of CBAs allowed by ESVs. At the micro level, much of the loss of biodiversity is due to development projects and strategies that are unsustainable both from a biodiversity and from an economic point of view but strive to push through the decision-making process anyway, because they are advantageous to some involved parties.

Improving the biodiversity situation involves making changes in development tracks. Sometimes small changes are enough. In other cases, large redirections of development paths are necessary. But in all cases, the impact of ESV depends on the overall success of mobilisation in favour of ecosystem services being taken into account in a decision. This mobilisation may take extremely diverse forms, being based on policy expertise or on advocacy, on militant campaigning, on lobbying, litigation, implementing safeguards in large organisations, on environmental mediation or collaborative processes and so forth. It is important to establish in each case what the decisive action course is, or may be, and where ESV could fit in it, alongside with a critical analysis of the overall economics of the case. After all, economists involved in ESV are both economists... and participants in collective action in favour of biodiversity.

10.2.3. *Factoring active resistance to change in the use of economic instruments*

As we now turn to "money on the table" economic tools – PES, buying land or land-based rights and biodiversity banking – strategic interaction moves from the realm of advocacy to the sphere of action. There is no doubt that by making payments, one of development projects, programs and strategies can often make things and people change course. What is more subject to discussion is the relation (often, the frank disconnection) between the discourse that accompanies the payments and the reality of how they work and of the respective intentions of the payer and the payee.

[35] Subsidies that reduce production costs in natural resource industries, thus encouraging resource depletion by artificially lowering extraction and production costs, often also maintain afloat activities of little or no economic value.

A major aspect of strategic analysis is to look beyond justificatory discourse at the strategic games that are going on between the actors. Here, we will examine economic instruments through some organising themes of the SEMA framework, starting with the fact that change in favour of biodiversity is the result of interplay between environmental actors' strategies for change, and the resistance of some other actors to environmentally motivated change. How does such resistance play out in the use of economic instruments?

At the stage of putting in place an ETB scheme, reasoning about how instruments work and comparing various possible instruments often implicitly assumes that there is an institution that has the power to use the tool to manipulate the behaviour of economic agents. An exception is the branch of economics that deal with principal-agent relations, as they examine how agents can manipulate information in return to avoid being steered by the principal where they would not like to go. Real political processes, however, differ from such a theoretical blueprint in two important ways. Economic agents do not act separately in their confrontation with government, but they are grouped in networks and organisations whose object is precisely to influence government. They do not only use information to that effect but are able to muster a whole arsenal of power resources to influence decisions. Just consider successive reforms of the European common agricultural policy. A model based on the idea of the EU using instruments to orient the behaviour of farmers and of farmers using information strategically to counteract such orientations would dramatically underplay the highly political and socially highly-charged process that is involved.

So if we consider a given economic tool as a way for an actor to manipulate the behaviour of another, if (1) an actor or organisation of actors does not want to (or cannot – in strategic terms, the boundary between the two is very fuzzy) change a given technology or production system and (2) it has sufficient power to prevent the creation of mechanisms that would press for such change, then it is highly likely that the mechanism will not be put in place – or that it will be watered down, often to the point of having no, or just marginal, effects. This sets limits to the usability of economic tools to cases where (a) an economic sector is willing to change its activities in a certain direction, and the economic tool would make such change more attractive or easier to implement or where (b) an economic sector cannot muster enough power to stop a mechanism being put in place, relative to the political or social pressure in favour of the mechanism. Certainly there are some cases in which an economic sector is almost ready to accept change, or is not very motivated to oppose it, so that an economic tool providing some advantages (through payments, or simplified procedures in biodiversity banking) may tip the balance and trigger a change of course. But this seems to us far from being the most general case. The important point here is that the situation one is in cannot be presumed, but should be precisely diagnosed in each case through a strategic analysis that factors in the whole power strategy of actors and (overt or covert) adversarial behaviour in the interactions about putting in place an ETB mechanism.

The capacity of the manipulee to manipulate the manipulator, as it were, does not stop at the stage of creating an ETB mechanism: it continues to reign at the implementation stage. This is particularly evident in the "eco-opportunistic" process we discussed about PES in chapter two, whereby a sector-based actor accepts payments but manages to dispense with the environmental conditions that initially motivated the payment scheme. This possibility exists as soon as the payer may lack the power to make sure conditions are met, whether the payer is a private or public operator. In PES, the consequence is payments without adequate provision of the ecosystem service. In buying land, it may lead to land (or land-based rights) being owned for conservation, but management agreements, or the enforcement of management agreements being such that the expected results are not there. In biodiversity banking, it may come in the form of inadequate offset through the manipulation of equivalences of ecological value between the land that is impacted and the offset proposed.

In brief, strategic resistance to the creation and to the implementation of ETBs should be quite high on the agenda of diagnosing situations for practical use. It should also be factored into the current debates on the generalisation and upscaling of ETBs. There is no reason that the strategic situations of opponents to a given mechanism should be the same at the small scale of pilot projects and in large-scale design and implementation. In some of the examples we gave in the four first chapters, some cases clearly indicated that an innovative and successful mechanism was tolerated by sector-based actors precisely because they were not to be extended to a larger scale. The Vittel case is particularly explicit, as the agreement between the water plant and the farmers was only tolerated by the farming industry and the Ministry of Agriculture on the explicit condition that it would have no spillover effect on other areas. There may, however, be other cases in which what is difficult at small scale may become more practicable at large scale, for instance when a national forestry organisation opposes a certain innovative local scheme to avoid setting a precedent: its resistance may not be much stronger if the change were promoted on a large scale at which, however, it might be possible to gather more support in favour of change than just on a local project.

10.2.4. *ETBs can strengthen the environmental sector*

To overcome resistance to change, ETBs have to be used by an environmental actor with enough strength, skill and expertise. Just as the users he is facing, the environmental actor does not act in isolation, but is himself part of a wider network of action, which we can call the biodiversity conservation sector. Just like the forestry, the farming or the mining sectors, it is a source of strength through union and of competence through the complementarity of statuses and skills – for instance, between environmental administrations, specialised research centres, NGOs, training programs, etc. One of the results of our review of ETBs was that their use almost invariably required the combination of economics and economic tools with equally high skill and investment in legal, administrative, scientific, sociological and other approaches of the biodiversity issue at hand. This complemenarity of approaches around shared purposes is precisely one of the benefits of activity sectors at all levels (from local to global) and at all stages from training to project implementation. To work in synergy with the other types of tools and approaches required in dealing with biodiversity issues, and to benefit from sufficient clout to take effect in the face of resistance to environmentally motivated change, the use of ETBs has to be embedded in a strong biodiversity sector.

Consider for instance payments for ecosystem services and, as we saw, their use of "deals" agreed to in negotiation. PES brings one important lever in such negotiations: the payments that can condition an agreement for change and help in making the change stick over time. But it provides no substitute for many other aspects of such negotiations. To succeed in such negotiations, the environmental actor using the tools requires sufficient clout and skills. For an NGO, this may imply for instance being backed by a governmental environmental agency, being familiar with the local field conditions through continued presence, being able to mobilise researchers, experts and interns in support, being able to disseminate information about the case in the specialised press or on specialised websites to help increase support from the public, being able to share experience and advice with others in the biodiversity sector, etc. Again, as he negotiates with actors who – apart from a few exceptions – are themselves backed (and often, actually constrained in terms of their margins of manoeuvre) by powerful activity sectors, the environmental actor who is the main "payer" in PES usually needs to find similar support in his own, environmental, sector. The same holds true for buying land or land-based rights: the means to buy, the level of support required to acquire the necessary goodwill, the expertise to choose and justify the purchases and the skill to implement it all quite rarely come to isolated operators. As for biodiversity banking, we showed in chapter four that of all ETBs, it is the one that relies most on strong, multifaceted support in terms of specialised law, administration and expertise.

Not only does the sound use of tools like ETB demand the resources of a sufficiently strong biodiversity sector, it is also worth considering to what extent their use is able to strengthen that sector. In that respect, the most interesting tools are buying land and biodiversity banking. Buying land (and land-based rights) not only solves individual biodiversity problems. Over time, and across places, it accumulates a growing estate. With it comes a growing level of influence, a growing organisation for supervising management, with the associated level of expertise, and also probably the means for further purchases. The presence of such stable and growing operators does strengthen the biodiversity sector. Biodiversity banking also provides such benefits, although in a different form. It fosters the creation of specialised businesses (or branches of businesses) to operate the banking. With them come specialised jobs, high levels of expertise of all kinds, and increased financial power in an activity that has biodiversity as an end. The high level of administration required also presupposes a strong administrative sector in the biodiversity field. But maybe the most striking feature of biodiversity banking in that respect is that if one envisages its application at large scales, it provides the basis for the growth of ecological engineering and of forces in the biodiversity sector that will reach beyond conservation efforts to increase ecological restoration and reconfiguration efforts which we think will also be necessary considering global change issues and the scale of the biodiversity crisis.

From a strategic perspective, the potential of ETBs for to strengthen those who act for biodiversity and the entire environmental sector is an essential consideration. This insistence by the SEMA framework on building up the strength of the environmental actor and on the sector-based dynamics involved in treating biodiversity issues may seem trivial to some practitioners in the field, whose experience it corroborates. It is surprising, however, how little attention it receives in the literature on biodiversity. At best, the needs of the biodiversity sector in terms of organised power are ignored (as if goodwill from all in favour of the cause were a given) or kept implicit. At worst, the very principle of actors defending biodiversity is criticised *per se*. This can be on the basis of a collaborative management view, in the fear that preparing for adversarial encounters would exacerbate antagonism and end up being counterproductive. Or, quite often, it is also on the basis of the defence of groups (for instance, local farmers) who are engaged in difficult negotiations with environmental actors and would prefer them not to be able to exercise any significant amount of power. The systematic critique of academics in the growing field of "environmental politics" against any sort of power backing environmental policy or advocacy provides a sort of generalisation of such reasoning. In a strategic environmental management analysis perspective, good management is the result of struggle and robust negotiation. And negotiation works only inasmuch as a party has enough power, of one sort or another, to press another out of the *status quo*, or away from its (biodiversity-wise) negative projects.

Conclusion

Roald Dahl's classic definition of power: "A has power over B to the extent that he can get B to do something that B would not otherwise do" provides food for thought for ETB users. Economic instruments are promoted precisely because they are based on the hope that they will allow to get people to bring about changes in activities that impact biodiversity. If such changes had been made anyway, then the usefulness of the tools is questionable. If the tools do bring about such change, then they involve a power relation. The important point here is that power is comprehensive: all aspects of the relation are involved. Economic incentives are included in, but they cannot be separated from, all the other aspects of the power relation: identity, organisation, leadership, politics, etc.

The SEMA framework helps focus analysis on the power relation between those who act for a given biodiversity stake and other actors from whom they have to obtain change if the biodiversity problem at stake is to be resolved. In this context, ETBs are useful inasmuch as they can provide more power for environmentally motivated change. Indeed, they can. Valuation generates powerful evidence and arguments; the money at stake in tools like PES or biodiversity banking does provide a tangible capacity to influence. ETBs generate power to act, to an extent. For real use of ETBs, it is essential to assess precisely that extent, based on a detailed diagnostic of the power games of each biodiversity management situation. It is essential also to be aware that any tool that has potential impact – i.e., power – becomes *ipso facto* the object of a power struggle to control the use of the tool. In a SEMA perspective, who uses the tools – a biodiversity-motivated actor, a sectoral stakeholding organisation potentially affected by biodiversity-motivated change, or a regulating actor trying to strike a compromise – make a decisive difference.

This entails stepping out of the economist's core competence. However, there remain some important continuities between the economist's approach and SEMA's strategic perspective on biodiversity management, so that the latter may still feel sufficiently familiar to many economists. On the level of principles, SEMA's model of biodiversity policy and management includes no grand design. It does not assume that anyone out there has either some overhanging competence or some overwhelming power that would allow them to design and implement a planned management of space and resources that would integrate all issues. It does not imagine – to paraphrase Crozier's phrase, that one could "change society by decree" (Crozier, 1988). It simply assumes that, somewhat like economies construct themselves from the interactions of agents – with some regulation added – real-life management of biodiversity is constructed through the strategic interaction of actors. This view of management as an open-ended and emergent rather than planned process, associated with the care taken of continuously clarifying the bottom line of that emergent management (in SEMA, the biodiversity bottom-line involved in the "actual management" concept) constitutes common-ground with economics. On a more concrete level, economists who care about the practical applications of economics have a practice of widening their scope to embrace strategic contexts and issues. Economists working on economic strategies at national scales would not envisage ignoring national politics, international relations and geopolitics. It should be only natural that economists working for better biodiversity management also include in their reflection the corresponding strategic contextual factors. Economists engaging issues of national economic strategies can rely on abundant scholarship, *e.g.* from political science or history, and on a common culture of such issues, widely disseminated by the media for instance. It is now time for environmental and ecological economists, in their effort to establish such a shared culture of strategic context, to mobilise similar resources, as they are being developed in research on the strategic analysis of biodiversity issues.

Conclusion

Economic tools (or instruments) for biodiversity (ETBs) are just what the phrase says: tools (or instruments). By themselves, they offer no guarantee of success. Success depends essentially on the tradesman (*e.g.* his capacity for diagnostic, his skill in choosing and using instruments), on the operative chain of action he follows, and on how that intervention fits the problem and its context. For good intervention design, implementation or evaluation, the characteristics of each tool, and the skill with which it is used, have to be set in the context of the entire trade that the tool is intended to serve. There is a need to rebalance the current treatment of ETBs by focusing on use and outcomes of the tools. In this book, we have set out to contribute to that effort by (a) reviewing the state of the art of ETBs use, and discussing the various issues met in the actual use of ETBs in context (chapters 1 to 5) and (b) laying out theoretical resources that can be mobilised to improve our understanding of how ETBs work in the actual contexts of biodiversity management and policy.

This final review of our findings will be presented in three parts: (1) an overview of the promises and limits of ETBs, (2) a synthesis of recommendations for practice and (3) some orientations for (much needed) further research on ETB use.

The promises and limits of ETBs

ETBs: panaceas, scarecrows or ghosts?

Are economic tools a panacea that will get us over the current impasses of the biodiversity crisis? Are they a threat, menacing to massively commercialise biodiversity issues, without really sorting them out? Or are they yet another soon-to-be-dispelled promise that we shall at last get down to business on biodiversity issues?

It is interesting to note that the vivid current critical debate around ETBs tends to polarise around the assumption that the tools are rapidly spreading and likely to become generalised, and on the idea that they would obey (and boost in return) a pure market logic. Reviewing actual use of the tools shows both assumptions to be quite far from the actual situation and dynamics of the development of ETBs. Our overall impression from examining the situation on the ground is that beyond pilot projects, ETBs' presence in actual practice is quite limited, apart from buying land for conservation and some PES schemes, most of them on water issues. Regarding some overenthusiastic ETB promoters, this may be a reminder of a basic fact about assessing the potential of a tool. If one focuses on the principle of a tool and abstracts it from all the contingencies of actual use, it is easy to find great potential in about any tool or gadget. Serious assessment of a tool's potential starts once you factor back in all the contingencies that the tool is going to meet in actual use. As we have seen, they are numerous and their impact on practice is high. This is also important for the more radical critiques of ETBs, as it means that the threats they rightly point to may usefully be qualified by putting them in realistic proportion with the real-world deployment of ETBs.

Our proposal, so as to move forward in the debate around ETBs, is to renounce reasoning that pushes deployment hypotheses to their limit, and to reason *in concreto,* in a pragmatic perspective. As we have seen, many of the *a priori* potentials and limits as identified by promoters and critiques remain interesting and useful in that context, but they take on different proportions and assume different meanings in view of concrete management and policy contexts. The question "What could, what did, happen in this precise, concrete management or policy situation?" is a great filter for screening advocacy and critiques of ETBs.

Another way to put positions in the ETB debate in perspective is to realise that the current biodiversity crisis vastly transcends the question of tools. What is intriguing is rather why we are so easily inclined to believe that a focus on tools will do the trick. Several hypotheses are possible. Maybe some reexamine the whole world from the keyhole of their favourite instrument. Or maybe others do not in fact care that much about the end results in terms of biodiversity, so concentrating on the benefits accruing from projects and instruments in themselves is fine with them. Others still may be in the grips of emergency, and consider that taking any action is better than sitting there and considering the complexity of things. In our view it is important never to lose sight of how the tools relate to specific biodiversity problems and the conditions that have to be met for these problems to be solved. This may bring us way beyond the scope of economic tools.

Probing the enduring perplexity about biodiversity issues

In our effort to recontextualise the use of ETBs, as we move from tool to trade, it is clear that the trade of managing biodiversity sustainably is in no way straightforward and raises deeper issues. An important one is the lack of a clear shared awareness of biodiversity issues, a pervasive perplexity that is, for many actors, an integral component of the issue. Again and again, biodiversity experts describe how they struggle trying to get operators to understand what biodiversity is and why it may be an issue for them. This confirms findings from other studies and from common experience of practitioners of the field. Indeed, part of it comes from the confusion generated by the term biodiversity itself, as it has replaced established and more tangible concepts and concerns like species, habitats, conservation, and as it has gradually extended its perimeter to cover many issues previously well identified and covered by other concepts in ecology. But such levels of perplexity are not caused just by terminological manoeuvres.

Most of the issues in biodiversity are still seen as new by so many actors, whereas they were already massively made public in the 1970s (as any ecology textbook of the time will show, for instance). The main doctrines inspiring policy have hardly changed since the 1980 IUCN conservation strategy that proposed sustainable development and life-sustaining ecosystems (an earlier formulation of the ecosystem services logic) as organising concepts. That in this context perplexity remains so widely present, so acute, deserves all our attention and is an integral part of the problem of biodiversity and of ETB use. How is one to reason on the use of tools... to solve a problem the existence or the nature of which raise such scepticism?

Three of the four theories we mobilised shed an interesting light on the fact that perplexity is an integral part of the processes of biodiversity management and policy. We take them here as an example of how using explicit alternative conceptual frameworks can help grasp more clearly and investigate in more detail specific pitfalls in the use of ETBs. Perplexity is most explicitly addressed by Bruno Latour (chapter 9) as he uses the word to describe the initial trouble that reigns, once resistance to taking a new problem into account has weakened, but serious negotiation on how to solve it is not quite yet underway. This author's works describes in minute detail how scientific uncertainty and its treatment are not separate from political arenas, debates and decisions – in biodiversity, the resolution of uncertainty should not be considered as a necessary preamble and condition for management, but as one dimension of the process in parallel with others. Justification theory (chapter 8) shows how in their interactions operators need to be able to refer to established orders of values to justify their claims or their decisions. Biodiversity is in the very awkward position of being able to be taken partly in charge by several of the prevailing order of values. But, as a result, it is also taken in the tensions that exist between them. And part of it also escapes the logics of these orders of value and will need the consolidation of a new ecological order of values in addition to the existing ones. Biodiversity concerns are also in part based on non-utilitarian, very divisive values in a way that conflicts with the massively dominating discourse that would have us all be in the same boat. Finally, strategic environmental management analysis (chapter 10) shows how both uncertainties and differences in values are exploited strategically by those whose activities would be threatened by better management of biodiversity. Perplexity is also the product of strategies based on fabricated "scepticism", strategies that are an integral part of environmental management processes.

Another aspect of the perplexity that we have found still prevailing is the difficulty to perceive, or to accept, fundamental differences between biodiversity and issues like climate change, pollution, water quality and water resources, renewable living resources (forestry, fisheries). All these can, and are to be managed (not exclusively, but largely) in terms of mastering flows and stocks – a paradigm that is common to engineers, most hard scientists, economists, and macro-scale managers, be it of the private or the public sector. Biodiversity in contrast relies in good part on the diversity of unique entities and patterns (species, habitats, place-dependant functioning, social-ecological arrangements, etc.) that cannot be grasped, nor managed, just by rationalising flows and stocks. Indeed, solutions that rely on the rationalisation and intensification of flows (for instance, biofuels, reforestation through plantations of best performing species) are themselves major causes of further biodiversity decline. This is a major issue, both practical and conceptual, that will be at the centre of all future perspectives in the success or failure of developing strategies and tools for biodiversity. For ETBs, the test is and will be their capacity not to foster uniformisation.

One of the interesting results of our work is to see that one of the roles ETBs currently play –the major one for ESV – is precisely to contribute to the working-out of perplexity: advocacy, awareness raising, translating biodiversity issues in various languages (economics being one of them) to help find a place for it in the babel tower of our complex societies.

Reaching limits, setting limits

Another recurring theme in our material points to the very wide context of biodiversity management and policy: the questions of the limits to development, to the intensification and extension of production systems that replace biodiversity-rich uses – or non-uses – of space. This combined intensification and extension is, at macro scale, the fundamental driver of biodiversity loss. A key question is: are we ready to set limits to the development of biodiversity damaging behaviour, production systems and industries?

Raised in the 1960s, the question of limits to exploitation of natural resources and pressure on ecological systems comes back with a vengeance after having been denied for decades.

When one examines the use of ETBs in detail, issues of limits come in many forms. If we valuate biodiversity, to what extent shall we be ready to renounce development projects when incorporating biodiversity valuation flips the calculus to show a theoretical lack of profitability (even if tangible, money-in-the-pocket profitability is still very much there for the project operators)? To what extent are we ready to alter property rights through buying land, easements and concessions to the point where it will limit surfaces available for further extension of intensive production? Are we prepared to launch ecological restoration and engineering programs that will be on a scale large enough to compensate for coming biodiversity degradation to come, knowing that such programs will inevitably limit the areas available for biodiversity-damaging development?

So the question of how, at all scales, we are prepared to limit not development as a whole, but some very concrete and important aspects of development, like the areas of land used for tilled agricultural production, or to plantation forests – is an essential dimension of the context for the use of ETBs. As tools, they will have potential for action on a large scale only where society is prepared to set some limits to development on the same scale. If ETBs are levers, they need a fulcrum, and the fulcrum is our actual determination and power to set such limits. If that power is weak and only local, ETBs will be tools to conserve or restore some biodiversity locally, while much more will continue to be destroyed. The scale and effect of all biodiversity management and policy tools will be determined by the scale and effectiveness of our more general setting of limits to biodiversity destruction.

Here, use-in-context of ETBs converges again with their economics background, which is the optimisation of the use of scarce resources. Setting clear limits to development that depletes biodiversity creates scarcities in the management of which ETBs can then be useful – as demonstrated for instance by the discussion of offsets and biodiversity banking in chapter 4. If we are willing and able to set limits to biodiversity depletion, economic valuation and instruments for biodiversity can help make such limits easier to manage, but they offer no easy solution out of the necessity to politically set and implement them. The various theoretical frameworks we discussed in the book each propose a relevant perspective on how we actually can set and implement limits to biodiversity-damaging activities, *e.g.* by laying down agreed-upon rules and institutions (chapter 7), by a clearer discussion of values (chapter 8), by innovation and social transformation (chapter 9) or by strategically confronting damaging activities (chapter 10). Each perspective allows us to acquire a more precise understanding of the chains of operations that ETBs are a tool for and that will eventually determine the usefulness of ETBs or otherwise.

A summary of our findings to guide the use of economic tools for biodiversity

Once ETBs have been put in perspective relative to the deeper issues of biodiversity management and policy, let us turn to the use of the tools itself. We shall assume that the operator is a fine tradesman – *i.e.* is versed in the various aspects of managing biodiversity issues and wants to add ETBs to his panoply and put them to good use.

No need for a new "economic tools" toolbox: filling in the slots of the biodiversity management toolbox will do!

A first step is to place economic tools in relation to the many other tools in the biodiversity management and policy toolbox.

If only by using the generic term "economic tools", one tends to set them apart in a specific slot, as if they had their own logic, quite different from that of the other tools. They do have points in common: the use of money as a language or as the operative principle of action; the scholarly presence of economists as experts. But our review of ETBs in the first four chapters has allowed us to measure the vast differences that exist between the various tools. In terms of what they are good for, and of the problems met in use, there is not that much in common – and in practice, even less than textbooks would suggest – between ESV and PES, PES and the buying of land or biodiversity banking.

Conversely, we have found each of the various sorts of ETBs to be quite close to some of the other items in the biodiversity toolbox.

Ecosystem services valuation (chapter 1) are akin to inventories, indicators and ecological evaluations. They rest on similar identification and qualification (of ES for instance) as inventories. They could be described as a particular kind (monetary) of indicator or evaluation. The kinds of usage they are good for is about the same: providing organised and hierarchised information for consideration by decision-makers (or participants in the decision-making process). The problems met in use are the same. The illusion that a good indicators system , like a good compass, would allow the ship of environmental management to be wisely steered is prevalent for indicators just as it is for ESV. Their actual main use as means for advocacy and justification is parallel. The apparent obviousness of the concept, as contrasted with the intricacies of implementation and the host of methodological problems that are met in practical use are also quite similar. In brief, both for practical use and for guiding the improvement of tools, ESV would best fit in the "information-for-management-tools" slot of the biodiversity tool rack.

Payment for ecosystem services (chapter 2) is part of the "contractual and incentives" set of tools. It sits for instance in the company of subsidies for clean technologies or regulatory bonuses for users (for instance, the right to enter limited traffic zones if you are driving an electric car). The fundamental logic of these tools is negotiated steering, the fundamental criterion being the fact that supplying is voluntary, not compulsory. This is a very flexible and adaptable set of tools. Many of the tools in that rack would count as "economic tools". Such incentive-based tools are in great use, but they raise questions of transparency (because they rapidly end up generating ad hoc systems of rules) and of effectiveness and cost (because of mechanisms like eco-opportunism or the manipulation of the principal by the agent that we discussed in chapter 2).

Tools based on ***buying land or land-based rights*** (chapter 3) are closely akin to protected areas. The difference may seem large at first sight, since some are based on acquisition, others on regulation. But both tools converge on what is really accomplished: the close control of given stretches of land (by ownership in one case, by regulatory means in the other). The control can be extensive (as in full ownership or in nature reserves) or it can be partial and specific (like in easements or targeted land-use regulations). The aims, the management patterns and the resulting challenges in practice are in large part similar. Another important common feature is the change that occurs between the strategic sequence that leads to the acquisition (or designation) of land, and the following sequence dominated by the concerns of long term management of land that has been marked for conservation. Overall, these tools fit in the "controlling specific areas for conservation" slot of the biodiversity tradesman rack.

Finally, ***offset and biodiversity banking*** (chapter 4) are an extension of EIA-based permitting systems and procedures. Not only can they only function if such procedures are in place and enforced rigorously, but they can be operated only as an integral part of the permitting procedure. They push the procedure one (offset) or two (banking, with the trade of offsets added) steps further but they don't fundamentally alter its nature. The users of these tools are all operators who are already engaged in EIA-based permitting: mining companies, developers, infrastructure-builders; their administrative counterparts; the several sorts of experts that have to intervene for ecological, legal, technical expertise. The problems of use and conditions for effective use are similar: they are essentially a combination of sound expertise and firm, carefully implemented, permitting procedures. The "identify, avoid, mitigate and compensate impacts" slot of the biodiversity toolbox is where that tool fits best.

This re-ordering of economic tools for biodiversity alongside with the rest of the toolbox suggests (1) that there is much less of a discontinuity between them and other biodiversity (and to a large extent, environmental) management tools and (2) that for the users of each tool, the problems met will be similar to problems met with other tools they are familiar with. Unpacking the "economic tools" box and putting ETBs away alongside other tools in the "biodiversity management and policy" toolrack moves them closer to the other tools they will be combined with, to the specific operations they will be used for and to the actual potential and challenges of their use for management and policy.

Improving diagnostics of biodiversity management problems/situations

Once the toolbox is well organised, it is time to turn to the problem, or situation that the tradesman has to face. It has been the book's leitmotiv that the use of an economic tool must be preceded by a relevant diagnostic of the management situation at hand. And much of the book has been devoted to providing concepts and frameworks to guide such diagnostics. Let us summarise the main findings in that respect.

Clarifying agency is of the essence (chapter 6) when planning for action on a biodiversity problem. The first thing about using a tool is to be very clear about who the operator is going to be, and about who is accountable to whom in resolving the problem at hand. In most of the ETB literature, this question is dealt with in a fuzzy way, either (a) by supposing implicitly that existing organisations are up to the task, (b) by imagining some sort of notional subject (for instance, the one who sets the incentives right) without checking that in reality someone has the corresponding power, or (c) by setting us all ("humankind") up as grand managers of biodiversity problems. For serious use of ETBs, it is necessary to use much more precise notions of who is acting for the common good and who is accountable to whom in that respect. The five paradigms we proposed are a good place to start for guiding such a diagnostic. They clarify the sometimes contradictory underlying models of agency we use sometimes when discussing use of ETBs. Confronted with a biodiversity problem/situation, they invite us to form a clearer concept of organised action on which to base our diagnostic and proposals for action.

The theories discussed in the last four chapters of the book can each help form a deeper understanding of organised action for biodiversity, and how ETBs can be used in such action.

Common-pool resources theory (chapter 7) puts at the centre of diagnostic the conditions that may allow successful negotiation of rules for the shared management of a common good. It shares the fundamental structure of the economist's analysis of resource management situations but allows its extensions into social, political and cultural dimensions.

Justification theory (chapter 8) helps in the diagnostic of the contradictory – but also sometimes complementary – normative orders of worth that underpin the value issues that are so central in the management of biodiversity issues. This is particularly useful in view (1) of the intense controversies triggered by ETBs and (2) of the difficulties so often met in trying to ground biodiversity conservation in values strong enough to withstand the enormous pressure of competing priorities.

Actor-network theory (chapter 9) focuses our attention on the joint transformations of knowledge, technology, power and actors, as social and ecological systems constantly reconfigure themselves. It puts institutional and technical innovation at the centre of diagnostic, and provides interesting concepts to grasp the innovation process and the hurdles innovation has to overcome. It helps us understand how in biodiversity management situations we gradually reconfigure jointly our priorities, our political relations, the techniques we use, and the way we understand our ecology.

Strategic environmental management analysis (chapter 10) orients diagnostic towards the power issues that are at play in biodiversity problems. It invites us to realise that in most situations, the user of the tool is in effect in charge of a campaign for biodiversity-motivated change. This involves a confrontation – usually in part an adversarial one – with other actors who are not so keen on such change, and requires a strategic diagnostic to identify margins of manoeuvre in that confrontation. That diagnostic will help ascertain how the tools can help in the confrontation of powers, or how the way that confrontation plays out in a given case can prevent ETBs from producing the outcomes we might otherwise have expected from them.

Strengthening the biodiversity sector

Finally we would like to underline one particular finding: the crucial role of intermediaries. From valuation (chapter 1) to biodiversity banking (chapter 4), all our observations converge to show that relevant and large-scale ETB use requires a strengthening of the professions that are needed to put the tools to good effect. ETBs create large needs in terms of science and expertise in ecology, but also in law and administration, in economics and finance. It requires entrepreneurs, brokers, professionals. In a word, tools need tradesmen and their impact will be dependent on the force and skill of the trade that uses them. It also requires sufficiently strong organisations, with adequate financial resources, as exist in all well-organised sectors of activity.

This ETB industry is itself nested in the wider sector of organisations and professions supporting biodiversity management and policy, from NGOs to specialised research centres, from environmental public agencies to specialised media, etc. At large scales, all crucial needs of societies – food, energy, water, forest products, transport, health, protection of the socially vulnerable… – rely on management and policy organised through specialised sectors of activity. As biodiversity becomes a major issue at all scales, organised action will have to be able to rely on a robust biodiversity sector. This is an important consideration both for management and policy. In deploying ETBs, one is at the same time funding the tools, supporting the tradesmen, and developing, organising and regulating the trade – because it is on the trade (*i.e.* on organised and capable users of tools for biodiversity conservation and management) that success ultimately relies.

The issue of financial needs for biodiversity, which is a significant aspect of ETBs, should also be discussed in this perspective of how the biodiversity sector can grow and evolve in the way it is organised to effectively play its role for conservation in the future.

This argument for the need of a strong specialised biodiversity sector – comparable for instance with the water management sector – is, however, a highly contentious issue. On a political level, the very sectors that are the main drivers of biodiversity loss (farming, forest industry, transport) claim that they are themselves the best managers of biodiversity and should be the main recipients of economic flows directed at biodiversity. As was discussed in chapter 2 and 10, there is a real strategic dilemma here: good biodiversity management depends on changes in the practice of sector-based actors involved in biodiversity loss and thus on some degree of collaboration of these sectors. But there are some deep connections between on the one hand the ways these sectors operate and how they are organised, and on the other hand the practices they promote and that impact negatively on biodiversity. Using tools for biodiversity then means in part using tools to change themselves, and this means inevitably some degree of external intervention. This is a major *raison d'être* for a specialised biodiversity sector and for a good part of the use of ETBs to be entrusted to it.

Sound strategies for biodiversity have to rely on a combination of strengthening the biodiversity sector proper, and of negotiating collaboration with sectors that impact biodiversity. ETB use can be part of one or the other sides – for instance, a conservancy that buys land strengthens the biodiversity sector, payments for ecosystem services through farming policies are centred on the farming sector. The decisive test for the second type of intervention is whether a given plan of action and the tools it uses is going to get adequate changes in the way the industry at stake behaves towards biodiversity, or whether it will mostly lead to increased funding, with only environmentally insignificant changes. Numerous case studies at all scales suggest that the limited success obtained so far in obtaining sufficient changes from biodiversity-impacting sectors is at the centre of the current limitations in our current ability to halt biodiversity loss. This is a key point of biodiversity strategies, from local to global. And who the use of ETBs is entrusted to often makes the difference between their productive use or their misuse, from the point of view of biodiversity effectiveness.

Perspectives for research

In this book, we have essentially tried to systematically lay down the problems raised by utilisation of ETBs in real-life biodiversity management and policy situations, to show that these problems would justify much more research than they attract now and to suggest some theoretical resources that may be useful both for reflective practice and for further research.

The most obvious perspective would be simply to intensify the effort for the documentation and analysis of cases of ETB use on the ground. Many of the topics covered in the book can help identify the questions such research may want to address. As what we are aiming at is understanding what makes management or policy succeed or fail, it is important to focus not so much on what economic theory has to say about tools, but on how the actors themselves interpret the tools and their use – and even more relevantly, what they actually do with the tools – and on how the actors and what they do relate in concreto with biodiversity goals. Of major importance is the necessity to break away from all kinds of assumptions that are taken for granted about how things work in management and policy. In fields like biodiversity, where so much that we do in fact does not really work, it is essential to reexamine how things really work (not how we superficially think they work). The challenge is compounded as we are dealing with biodiversity problems that are raised at several scales from local to global. For each scale we have widely shared implicit models of how things work or could work – the self-accountable community, the participatory planning at regional scale, rational and transparent national policies, the institutional construction of international regimes. These models of organisation and action that so pervade the state of the art fall far short of the complexity of action and of the requisite of adequate diagnostics to guide it.

Pushing beyond them requires explicit hypotheses, based on appropriate theoretical frameworks. After reviewing large sections of the ETB literature, we are convinced that only a fraction of the available resources are mobilised. This is why we have devoted the bulk of our effort to present several theoretical frameworks that we think can provide useful bases for further work, and to probe their potential for illuminating ETB use issues. We are well aware that as we have initiated several such tracks for investigation, we have not been able to go very far into each of the perspectives we propose. We nevertheless chose this approach for two reasons: (1) we thought that the current stage of research on ETB use is such that it was a priority to map out the terrain and (2) we are convinced that it is essential that a larger variety of different theoretical perspectives be used in the further development of such research. This could be perceived as an encouragement to an eclectic approach to the treatment of biodiversity management and policy situations, processes and strategies – an approach that would encourage researchers, students and practitioners to know a bit of everything in the social, managerial and political fields, and use it on an ad hoc basis.

On the contrary, we think that this belief (that an at-a-glance overview, or simple typologies, of context issues could be sufficient – a belief one sometimes finds reflected in the current literature on the use of ETBs), falls far short of the mark. The main message of the book on this point is that what is needed now is a new body of work, a diversity of case studies and theoretical research projects, each of which will adopt its own, clear, explicit perspective, well-equipped with theoretical and methodological resources, and investigate in depth some decisive aspect of biodiversity management and policy and the use of ETBs in that context. We have juxtaposed several perspectives here to demonstrate (1) that work has not gone far yet and that as soon as one mobilises a bit of theory relevant to use-in-context, there are insights to be found, (2) that there are many possible routes to deepen such work and that it is important that it should be carried out concurrently, by different people, on different routes and (3) to show that some perspectives can yield very valuable insights and diagnostic tools for the practitioner. Let us add that we have considered here only just a few of the potentially fruitful perspectives and resources. There are many more to be mobilised, and we hope that our provisional mapping will stimulate their identification and use. We have provided neither an overview nor a typology of perspectives and resources, but a sample of resources to encourage deeper use and further investigation.

Our final word will be on the relation between research on biodiversity management and policy and the concern for their effectiveness, *i.e.* for effectively tackling biodiversity problems. Many authors in the ETB field quite relevantly insist on the necessity of an interdisciplinary research effort to understand the multiple issues that are involved in the use of ETBs. Indeed, a number of disciplines have important contributions to make to our understanding of how economic tools are used in real-life social and political situations. Although we didn't insist on it, the various theoretical perspectives discussed here originate in distinct disciplines within social sciences – institutional and experimental economics, moral sociology, sociology of science, strategic management. Many others can be summoned, with further resources of their own. There is really a lot of room for a large variety of further research here. The one point we would like to draw attention to is that it is not quite the same thing to produce research about the use of tools in biodiversity management and policy, and research for the biodiversity-wise effective use of tools in biodiversity management and policy. Think of research on the firm. It is not the same to work on the sociology of industrial firms and to work on how to improve management of firms in view of their sociological dynamics. As we call for more research on the use of tools for biodiversity management and policy, we do hope that there will be more and more research on the use of these tools, but we also insist that a good part of it should be guided by the organising concern of effectively solving biodiversity problems.

Acronyms and Abbreviations

CAP	Common agricultural policy
CAS	Centre d'analyse stratégique (auprès du premier ministre)
CBA	Cost-benefit analysis
CPR	Common-pool resources
DPSIR	Drivers, pressures, states, impacts, response
EBC	Economic biodiversity credits
EC	European community
EIA	Environmental impact assessment
ES	Ecosystem services
EU	European union
ESV	Ecosystem services valuation
ETB	Economic tools for biodiversity
IAD	Institutional analysis and development framework
IDCP	Integrated development and conservation project
IDDRI	Institut du développement durable et des relations internationales
INRA	Institut national de la recherche agronomique (France)
ITQ	Individual transferable quotas
IUCN	International union for the conservation of nature
MEA	Millenium ecosystem assessment
MPI	Multy party institution
NGO	Non-governmental organisation

NSW	New South Wales (Australia)
PES	Payments for ecosystem services
PMPOA	Program for the limitation of pollution of animal origin
REDD	Reducing emissions from deforestation and forest degradation
SEMA	Strategic environmental management analysis
SPA	Social process approach
TEEB	The economics of ecosystems and biodiversity
TNC	The nature conservancy
UESV	Use of ecosystem services valuation
USD	US dollars
VRA	Value revealing approach

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Tools for what trade?

Analysing the Utilisation of Economic Instruments and Valuations in Biodiversity Management

Efforts to step up the use of economic valuation and economic policy instruments have gradually become a major theme of debate about policies to curb the biodiversity crisis. There seems to remain, however, a considerable gap between the extensive presence of economic tools in policy discourse on biodiversity and the limited level of use of the tools in the field. There is also a great discrepancy between theoretical justifications for the tools and how they actually operate on the ground. There is now a need to focus on the actual use, rather than on the principles, of economic tools for biodiversity. In this book, we contribute to this change of focus in two ways. On the one hand, we use the literature and interviews to systematically review economic tools for biodiversity to identify the specific issues raised by their use. On the other hand, we lay down a repertoire of theoretical resources that we think are particularly relevant to acquire an in-depth understanding of how these tools actually function in the real world of biodiversity management and policy.

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