Ecosystem Services and Institutional Dynamics

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Article

The Biodiversity Offsetting Dilemma: Between Economic Rationales and Ecological Dynamics

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Abstract: Although many countries have included biodiversity offsetting (BO) requirements in their environmental regulations over the past four decades, this mechanism has recently been the object of renewed political interest. Incorporated into the mitigation hierarchy in three steps aimed at avoiding, reducing and offsetting residual impacts on biodiversity arising from development projects, BO is promoted as the way to achieve the political goal of No Net Loss of biodiversity (NNL). The recent success of BO is mainly based on its ability to provide economic incentives for biodiversity conservation. However, the diversity of BO mechanisms (direct offsets, banking mechanism and offsetting funds) and the various institutional frameworks within which they are applied generate substantial confusion about their economic and ecological implications. In this article, we first analyze the rationale for the BO approach from the welfare and ecological economics. We show that both these frameworks support the use of BO to address environmental externalities, but that they differ in how they consider the substitutability issue and levels of sustainability with regard to natural and manufactured capital, and in how they address ecological concerns. We then examine the economic and ecological performance criteria of BO from conceptual and empirical perspectives. We highlight that the three BO mechanisms involve different economic and ecological logics and inherent benefits, but also potential risks in meeting biodiversity conservation targets. We lastly investigate the ecological constraints with respect to the BO practice, and economic and organizational limitations of the BO system that may impede achievement of NNL goals. We then reveal the existence of a tension between the economic and ecological rationales in conducting BO that requires making choices about the NNL policy objectives. Finally, this article questions the place of BO in conservation policies and discusses the trade-off between political will and ecological opportunities involved in the BO approach.

Keywords: biodiversity conservation; biodiversity offsets; ecological compensation; economic incentives; environmental policies; human well-being; natural capital; no net loss; substitutability; weak and strong sustainability

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1. Introduction

Over the last two decades, environmental policies have increasingly used economic incentives for biodiversity conservation as more efficient ways of achieving conservation outcomes than traditional approaches [1]. Seen as a way to provide economic incentives, the concept of biodiversity offsetting (henceforth BO) has recently enjoyed renewed political interest, and is endorsed in many political agendas [2]. Whilst BO requirements have been appearing in the environmental regulations of many countries since the 1970s (but rarely implemented in practice [3]), BO has recently re-emerged in biodiversity strategies across national and international policies as the main innovative tool for biodiversity conservation worldwide [4]. Embodying a regulatory requirement, BO is primarily incorporated by law into the mitigation hierarchy in three steps aimed at avoiding, reducing and offsetting residual impacts on biodiversity arising from development projects [5]. The purpose of BO is to provide ecological gains counterbalancing negative impacts on biodiversity. In a context of economic development, BO is considered the main way to achieve the goal of No Net Loss (henceforth NNL) of biodiversity, currently being a central political objective [6].

In practice, the BO principle encompasses three main mechanisms: (1) direct offsets, requiring developers to carry out compensatory measures themselves through restoration actions or acquisitions of natural areas in which appropriate conservation plans are implemented; (2) the banking mechanism, whereby a third party called a bank operator implements larger restoration projects ahead of future impacts, generating thereby offsetting credits for future needs of developers; and (3) offsetting funds, organized by certain environmental organizations (public agencies or non-governmental conservation organizations) in order to collect money from developers to carry out restoration actions or conservation projects [7]. How these different mechanisms are used and regulated depends on the legislation and the institutional environments of each country [8]. In addition to this, there are also voluntary offsets in which developers propose offsets outside legal requirements, but we do not propose to treat them in this paper.

The rationale for the BO approach is to achieve ecological outcomes in a more efficient way than in traditional political approaches, being commonly regarded as a market-based instrument (henceforth MBI) [9]. Yet, while BO is often regarded as an MBI for biodiversity both in academic and political spheres (especially in the form of the banking mechanism), some recent articles have shown that BO mechanisms do not really share the characteristics of market or market-like instruments, either in theory or in practice [10,11]. However, most academics and policy-makers still value BO on economic grounds, the revival of BO in environmental policies largely resting on its ability to promote economic incentives for biodiversity conservation. Overall, by allowing environmental outcomes to be achieved without limiting economic development, the concept of BO offers the promise of making economic development and growth compatible with biodiversity conservation [12].

In addition, the economic rationale behind the BO scheme has raised major concerns with both academics and conservationists especially with respect to ecological goals. For some, this approach gives rise to a commodification of biodiversity that tends to jeopardize biodiversity conservation instead of ensuring it [9,13]. These observations have raised questions about the limitations and risks of using the BO approach for biodiversity conservation. Recent reviews on BO emphasized the main conceptual and practical limitations involved in the implementation of offsets (e.g., specific problems associated with metrics, equivalence, timing, spatial, compliance, monitoring, *etc.*) [14,15].

However to date, the economic foundations of the BO concept have rarely been addressed in the scientific literature. Moreover, the diversity of the BO mechanisms and the various institutional frameworks within which they are applied generate substantial confusion about their economic and ecological implications. Far from being exclusively of academic interest, analyzing the economic rationales of the BO approach should yield a better understanding of the functioning of its mechanisms, and help in dealing with their economic and ecological limitations.

In this article, we conduct such an analysis in three parts. First we examine the overall economic foundations and rationales for the BO approach to biodiversity conservation from an economics perspective. Whilst it is possible to address the overall rationale behind the BO scheme, we assume that the performance criteria of BO will vary across the three different BO mechanisms. We address this issue in the second part in two complementary ways. First, from a conceptual perspective, we examine the performance criteria of BO according to NNL policy goals and the type of equivalence targeted. Then, from an empirical perspective, we conduct a systematic analysis of the three different BO mechanisms in three steps by first describing their functioning, secondly pointing out their economic and ecological benefits, and thirdly highlighting their main risks. Finally, in the third part of this article, we discuss the main economic and ecological structural limitations and challenges of the BO approach when it comes to meeting biodiversity conservation objectives.

2. Economic Foundations and Rationales for the BO Approach

The principal objective behind the BO approach is to maintain biodiversity so as to achieve NNL of biodiversity in contexts where biodiversity losses occurred from development projects. BO is thus primarily addressed through the legal framework of Environmental Impacts Assessment [16]. In order to obtain permits, and in compliance with the mitigation hierarchy, developers are required to take compensatory measures to offset their environmental impacts leading to provide equivalent biodiversity gains. In most countries, environmental regulations aimed at in-kind offsetting mainly targeting “like-for-like” equivalence. This means that offset projects are designed according to ecological outcomes through actions of restoration, rehabilitation, creation or preservation of species and ecosystems [7,17]. In this perspective, regulatory frameworks only take into account the ecological gains provided by BO projects, regardless of the social or economic impacts of biodiversity losses. Thus, the rationale for using the BO system is primarily ecological.

However from an economics perspective, biodiversity losses represent social costs that go beyond the purely ecological level and need to be taken into account. Concern about the costs of loss of biodiversity has increased in recent years, especially since the publication of the Millennium Ecosystem Assessment reports [18] and The Economics of Ecosystems and Biodiversity reports [19]. These studies provide global economic assessments of biodiversity and ecosystem services, offering a general framework to link biodiversity and human well-being. They help to recognize that biodiversity underpins human well-being. In such a way, environmental losses are regarded as negative externalities that represent major costs for society and tend to reduce human well-being [20]. In this perspective, the main problems highlighted are decision-makers' failure to take biodiversity losses into account in economic calculations, and the lack of policy tools encouraging the internalization of negative externalities resulting from biodiversity losses [19]. Following the finding that traditional approaches failed in meeting the expected conservation outcomes, economic incentives have increasingly been used by policy-makers over recent decades to address environmental concerns [2]. Incentive approach aims to encourage economic decision-makers to adopt good environmental practices by offering compensation or rewards to individuals in exchange for environmental services [12].

By combining a regulatory approach, through the polluter-pays principle, with an economic incentives structure, the BO approach gained increasingly credibility and interest in political spheres [21,22].

The BO approach, in addition to ensuring that the legal compensatory obligation is met, should provide three major economic incentives that will influence developers' behavior and encourage good environmental practices. First, because BO represents significant costs for developers, it should be an incentive for developers to limit their impacts on biodiversity. Based on the insight that rational actors will perfectly weigh the economic costs and benefits of making their choices, developers are expected to minimize offsets, thereby reducing the impacts on biodiversity from their development projects. Second, the economic rationale of developers should lead them to comply with their offsetting requirements in the most efficient way, by seeking effective conservation projects [17]. Thus, if BO implementation is well framed and controlled by regulatory bodies, developers should in turn implement the best environmental practices by choosing the most cost-effective way to meet their offsets requirements (e.g., by using biodiversity banks). Lastly, through the financial benefits provided by some BO mechanisms, the BO system may provide incentives for private or public stakeholders to invest in conservation actions for economic reasons. The BO system can therefore exploit additional sources of funding for conservation actions, and may open the way to large and expensive conservation projects that probably could not have been implemented otherwise [23].

From the legal perspective, BO basically allows the internalization of negative externalities by requiring developers to offset the environmental losses they are causing. However, the rationale for the offsetting approach and the kind of offsets required can be regarded in different ways depending on the economics frameworks considered, especially with regard to welfare economics and ecological economics [24]. The main difference between these frameworks lies in how they consider the degree of substitution between the different forms of capital, especially natural capital and manufactured capital [25], and the different kinds of sustainability they imply [26,27].

Welfare economics aims to find solutions to internalize negative externalities (resulting for instance from development projects) in order to maintain the level of social welfare. In the welfare economics framework, the hypothesis is one of weak sustainability, meaning that manufactured and natural capital can perfectly be substituted for each other: what matters for future generations is only the total aggregate stock of manufactured and natural capital, but not natural capital *per se* [26]. According to this viewpoint, there is high degree of substitution between manufactured and natural capital implying that natural capital may decrease as long as manufactured capital increases in accordance with maintaining human well-being. In this perspective, there is no reason to specifically preserve the natural capital [28,29]. However, what matters is to maintain the social welfare then expressed through the utility provided by the production of goods and services. Welfare theory is principally based on the Pareto-optimum principle which states "that social welfare is maximum when it is impossible to make anyone better off (*i.e.*, happier or in a preferred situation) without making someone else worse off (with initial endowments)" [30]. According to this principle, economic activity is limited because no development projects can be Pareto-optimal since they give rise to negative externalities and decrease the utility of at least one individual [31]. A solution to overcome this problem lies in the Kaldor-Hicks compensation principle specifying that "as long as the sum of the total benefits of the project is greater than the sum of the total costs, a share of the benefits can be devoted to offsetting the social costs to meet the Pareto-optimality condition" [32,33]. In such situation, the compensation principle states that a change (e.g., resulting from development projects) is socially desirable if the individuals who are gaining from the new situation provide offsets to those who are suffering the losses. Moreover, if the costs of the offsets are borne by the developers and included in the total cost of the development project, the offsetting mechanism enables the internalization of negative externalities. The level of the offsets needed is assessed on the basis of the expected losses of utility. If losses of utility are assessed in monetary terms, offsets could be provided by financial gains. In this case, losses of natural capital will therefore be offset by financial gains instead of ecological gains. Thus, even if the welfare framework supports the use of offsetting to maintain the social welfare, high degree of substitution between the different types of capital is assumed (including natural capital and manufactured capital) because losses of natural capital are replaced by gains in other forms of capital. In this case, this is not about "biodiversity" offsetting but rather "utility" offsetting. However, the weak sustainability hypothesis can be regarded as over-optimistic in the light of recent works especially the TEEB and MEA reports leading to a very paradoxical situation called "the paradox of the environmentalist" [34,35]. Whereas values of biodiversity have been shown and the need to preserve it to maintain human well-being, it might be possible for the society to fall into an irreversible and highly degraded state of biodiversity while human well-being continues to increase, at least in the short term. With regard to this situation, if social welfare is to be maintained, utility losses resulting from biodiversity losses must be offset by gains in biodiversity and not by financial gains. Therefore in this case, beyond the hypothesis of high substitutability between the different forms of capital, a strong sustainability approach should be considered even in the welfare economics framework [36].

Conversely to this framework, the ecological economics approach is basically based on the strong sustainability criterion. This approach assumes that natural capital is an essential production factor, thus considering that natural capital and manufactured capital are complementary and not perfectly substitutable. The strong sustainability perspective defines ecological sustainability as "the natural limits set by the carrying capacity of the natural environment (physically, chemically and biologically), so that human use does not irreversibly impair the integrity and proper functioning of its natural processes and components" [37]. According to this viewpoint, a decrease in natural capital cannot be compensated for by an increase in manufactured capital [38]. In this case, offsetting cannot be financial and must result in gains in natural capital to maintain its level. However, the strong sustainability approach raises the problem of choosing the critical natural capital to maintain, and the minimum threshold levels below which they must not fall [39].

In practice, the implementation of the BO principle in environmental regulations aimed at achieving the NNL objective takes the strong sustainability perspective. In the context of development projects, losses in natural capital must be offset by gains in natural capital, implying that natural capital and manufactured capital cannot be substituted for each other. Moreover, the setting of the NNL objective in environmental policies reveals public and political awareness of the need to preserve biodiversity. The NNL goal was set during George H.W. Bush's campaign in 1988 in the United States of America (USA), initially to limit wetlands destruction [40]. The NNL objective then spread throughout the world, more recently becoming a political principle endorsed by many countries [6]. This commitment reveals the recognition of the various values and merits of biodiversity (social, economic and ecological) and the importance of maintaining natural capital by preserving it. Moreover, in contexts of strong ecological uncertainties about current and future biodiversity states and changes, incomplete knowledge about optimal levels of biodiversity, and when ecological extinctions are difficult to forecast [41], setting the NNL objective then represents a precautionary approach. According to the NNL perspective, two social choices are possible: either it is decided not to destroy the biodiversity that needs to be maintained, or it is decided to continue the economic and social developments because of their utility despite their environmental damages, but in this case, compensations for the destruction of biodiversity are involved. In this perspective, the BO principle represents the only way to reach the NNL objective. However, the level and type of ecological equivalence required between gains and losses depend on the equivalence criterion set in the NNL policy. The strict like-for-like equivalence represents for instance the highest levels of sustainability under BO regulations [3].

However, depending on the goals targeted in the NNL policy, different types of equivalence and associated offsetting can be provided. Indeed, the types of equivalence targeted in the BO system are closely linked to the goals of the NNL policies [3]. In line with Quétier *et al.* (2014), we examine from a conceptual perspective the various types of equivalence linked to NNL goals, and the associated BO baseline and metrics used to assess losses and gains in biodiversity (Table 1). When the NNL goal aims to maintain the level of human well-being, equivalence is based on the utility provided by the different forms of capital. In this case, losses of natural capital can be offset by gains in another capital, *i.e.*, manufactured capital (line 1, Table 1). Monetary metrics can be used to assess losses and gains using cost-benefits analysis to assess benefits of development projects and costs of biodiversity losses. From this perspective, a weak sustainability approach is assumed as high degree of substitutability between capital (see section above).

Conversely, when losses of natural capital must be offset by gains in natural capital, BO requires an ecological equivalence based on a strong sustainability approach. However, we propose to better reflect the practices of BO policies by introducing a finer distinction between the different components of natural capital based on three main approaches to biodiversity that are: ecosystem services, functional, and individual with species and habitats [42] (lines 2 to 4, Table 1). We further argue that these different approaches to natural capital involve different types of ecological equivalence and BO approaches. NNL goal can relate to ecosystem services that need to be maintained to human well-being. In this case, offsetting aims to replace ecosystem services damages. Ecological indicators tied to the different categories of ecosystem services can be used to assess losses and gains in ecosystem services [43]. A functional approach to natural capital can also be considered, with BO aimed at maintaining NNL of ecological functions. Functional indicators can be used to assess losses and gains in ecological functions [44]. Lastly, through an NNL goal focused on an individual approach to biodiversity (*i.e.*, species or habitats), BO is aimed at maintaining populations or communities of species or specific habitats. Here, biological indicators can be used to assess species (vegetal or animal) losses and gains [15]. These different approaches to biodiversity involve different ways of looking at biodiversity in particular with regard to the recognition of the complexity of its dynamics and processes (see Section 4.1).

Table 1. Types of equivalence, offsetting and metrics across No Net Loss (NNL) policy goals.

	NNL Goals	Types of Equivalence	Types of Offsetting	Possible Metrics Used for Assessing Losses and Gains
1	Maintaining human well-being	Equivalence in utility	Forms of capital: losses in natural capital can be offset by gains in another capital (e.g., manufactured capital)	Benefits of development projects *versus* values of biodiversity losses assessed through cost-benefits analysis
2	Maintaining the level of ecosystem services that are beneficial to human well-being	Equivalence in ecosystem services	Offsetting aimed at maintaining the production of damaged ecosystem services by providing equivalent gains in ecosystem services	Ecological indicators of ecosystem services by category (regulation, support, provision, cultural) (e.g., presence of species providing specific ecosystem services)
3	Maintaining ecological functions	Functional equivalence	Losses of ecological functions are offset by gains in the same ecological functions (e.g., habitat for species)	Functional indicators (e.g., habitat area, density of vegetation)
4	Maintaining species and habitats	Individual-based equivalence	Offsetting aims to replace the same species populations or communities and habitats lost	Biological indicators (e.g., presence/absence, species diversity)

3. Economic and Ecological Analysis of the BO mechanisms Performance

NNL policies commonly target, in practice, functional and individual (species or habitats) approaches to biodiversity to conduct BO. As stated above, three different mechanisms can be used to implement BO: direct offsets, banking mechanism and offsetting funds. We argue that economic and ecological criteria performance will vary among these three mechanisms [45].

3.1. The Direct Offsets Approach

Direct offsets involve the implementation of BO by the party responsible for environmental damages arising from development projects. In the US legislation, this system is commonly called "the permittee-responsible mitigation". Offsets are single conservation projects tied to a given development project. In this way, each offsetting measure is defined and sized case by case in relation to specific quantified impacts [14]. Moreover, offsets are primarily carried out near the impacted area [7].

From an ecological perspective, this proximity between the offsetting site and the impacted area, combined with defining each offset in terms of one impact, should help reach the NNL objective through better ecological and geographical matching between biodiversity losses and gains [17]. However, for this direct offsets system to work, offsetting measures need to be properly incorporated into local conservation projects. Studies have shown that single offsets implemented regardless of local conservation projects and without being incorporated into spatial and temporal planning can lead to conservation failures [45]. Single offsets can also result in small and isolated conservation projects leading to ineffective conservation outcomes [46].

From the economic and organizational perspectives, the direct offsets approach is considered as being inefficient. In the USA, at the end of the 1980s, two reports pointed to shortcomings in the application of BO through the direct offsets approach leading to biodiversity losses [47,48]. These failures actually revealed major organizational difficulties in the enforcement of offsets requirements through the direct approach. This inefficiency was mainly due to the high transaction costs generated by the implementation of biodiversity offsets in individual cases both for regulatory bodies and developers especially where large development projects are concerned [49]. Indeed, in the direct offsets system, the developers themselves are required to implement their offsetting measures. However, they generally do not have the necessary skills to conduct offsets that require significant expertise and specific knowledge. In this case, developers generally use service providers and experts to conduct their offsets, but this increases the financial costs of offsets measures. However, the level of expertise needed is highly dependent on the NNL goals and on the type of compensatory measure targeted [50]. For instance, preservation measures (*i.e.*, purchasing an existing natural area in order to preserve it) require lower levels of expertise and knowledge than restoration or creation actions [50]. However, the implementation of BO requirements through preservation actions raises strong concerns among scholars and conservationists in relation to the issue of the additionality of compensatory measures to meet the NNL goal [14]. In the USA, since BO regulations were reinforced, restoration measures have accounted for the highest proportion of direct offsets measures carried out (42% [51]); in contrast, such measures account for the smallest proportion of offsets in France (17% [52]). For the regulatory bodies responsible for the enforcement and monitoring of BO, offsets carried out through the direct approach generate high transaction costs too. This approach requires regulatory bodies to control and monitor as many offset projects as development projects, which generates significant transaction costs and makes it difficult to enforce offsetting liabilities [49,50]. In the USA, this inefficiency of the direct offsets system led to the implementation of the banking mechanism in early 1990s in response to these economic and ecological flaws [13].

3.2. The BO Banking Mechanism

The BO banking is an innovative organizational form that emerged to meet offsetting requirements. Through this approach, a third part called an operator carries out offsetting measures on behalf of developers by creating an offsetting bank. This bank is composed of biodiversity credits corresponding to ecological gains provided by the bank operator. These gains generally result from restoration actions conducted ahead of future development projects that have been checked and approved by regulatory bodies before being used to offset developers' impacts. Thus, an offsetting bank serves to offset several and various impacts arising from different development projects. One of the main differences between this approach and the direct offsets approach is the transfer of responsibility from the developers to the bank operator in conducting and monitoring the compensatory measures over time (note that the period of liability depends on national regulations; under US law it is forever because conservation easements are linked to offsetting banks, while in France, the average period is about thirty years [53]). Legal responsibility for offsets can also be transferred to the bank operator, but this too depends on the regulatory framework (transfer is possible under the US law whereas in France the developer retains legal liability [53]). The banking mechanism actually encompasses various schemes under different names according to regulatory and institutional frameworks and to the type of biodiversity targeted (e.g., species or habitat bank, wetland mitigation bank, biobank). The bank operator can be a private or a public organization or individuals, and the offsetting bank can be commercial or non-commercial [7].

From an ecological perspective, the banking mechanism is supposed providing better conservation outcomes than in the direct offsets approach. First, by pooling various small offset actions within a larger offsetting project, the banking mechanism better guarantees ecological and conservation successes [54,55]. Combining the offsetting liabilities of several developers over a larger area providing greater ecological benefits increases the chances of successful offsets coming from biodiversity banks. Moreover, planning in advance for offsets through the banking mechanism encourages the right choice of offset sites and actions to be made in relation to local conservation issues. In addition, the banking approach helps prevent temporary losses of biodiversity by providing biodiversity gains ahead of future ecological impacts [56]. Indeed, a major argument for the ecological benefits provided by biodiversity banking is the effective temporal and spatial strategy that the mechanism encourages, in terms of offset locations and types (*i.e.*, ecological actions) [45]. Advance checking and approval by federal agencies of the ecological results that the bank proposes to offset future impacts also limits offsets failures [57]. Lastly, depending on the level of asset specificity set by regulations, the banking system can even aim for significant environmental gains, leading to good ecological restoration projects.

However, although the banking mechanism was expected to ensure ecological success, many case studies have revealed difficulties and failures in achieving NNL of biodiversity [58,59]. These findings have led to an extensive academic debate on the relevance of such tools for biodiversity conservation [60,61]. On technical concerns, some studies revealed particular problems in relation to spatial issues and restoration results [62,63]. Contrary to the direct offsets approach, the banking mechanism necessarily implies off-site offsets and greater gaps between losses and gains in biodiversity, despite the definition of a specific service area (*i.e.*, a geographic area in which the bank can sell its credits; in the US wetland mitigation banking, this is usually a sub-basin area from 255 km^2 to 3544 km^2 in size, depending on the State [64]). Indeed, this mechanism requires a sufficiently large area for the offsetting market to function properly (increasing offsets demand), which tends to reduce the geographical equivalence between ecological losses and gains. Moreover, due to the possibility of offsetting for multiple impacts via the same environmental gain, asset specificity is decreased [50]. Thus, as biodiversity offsets are not sized and carried out according to one specific ecological impact, the ecological equivalence between biodiversity losses and gains is supposed to be weaker than in direct offsets. In fact, for the banking mechanism to function properly, biodiversity credits need to be sufficiently standardized to be equivalent to several ecological impacts.

From the economic and organizational perspectives, the BO banking mechanism is commonly regarded as an MBI both in academic and political spheres (although some academics have challenged its implementation as such [64,65]). In theory, MBIs are expected to reach any desired level of ecological objectives in the most efficient way (*i.e.*, at the lowest cost) if they are properly designed and implemented [66]. In the case of BO, the desired level corresponds to the NNL objective that requires impacts on biodiversity to be offset. Due to specific features of the banking mechanism, this approach is expected to be the most efficient way to implement offsets [67]. The banking mechanism can be also regarded as an economic incentive for both developers and bank operators (see Section 2). Developers and bank operators are expected to find the most efficient ways to carry out offsets [66]. The main advantage of the banking mechanism lies in the use of an intermediary, which greatly reduces transaction costs for developers and regulatory bodies [49]. Through this mechanism, developers transfer the costs of implementation, management and monitoring of offsets to bank operators strongly limiting their transaction costs. Even though regulatory bodies spend time setting up an offsetting bank, once the bank is operational, less time and work are required to check and monitor offsets than with direct offsets, due to economies of scale. Thus, regulatory bodies can ensure better control and monitoring of offsets, leading to better enforcement of environmental regulations [50]. As for bank operators, even though they bear the transaction costs tied to offsets, they can achieve cost-effective implementation of offsets by taking advantage of the economies of scale resulting from large offsetting projects [68].

Turning briefly to the social dimension, the banking mechanism fosters the development of partnerships between parties who do not usually interact. Implementing an offsetting bank leads various stakeholders (public, private, organizations or individuals) from different sectors (business, conservation, agriculture) to exchange views on environmental issues. The banking mechanism encourages them to communicate and share, in order to balance their different goals. However, the banking mechanism also raises social inequality issues. The main concern is that those benefitting from offsets are not those suffering as a result of ecological losses. The banking mechanism implies a spatial gap between the impact and offsetting areas, meaning that the people who benefit from compensatory measures are not likely to be those who suffer from the environmental damage [69]. Although this issue deserves to be explored further, it is beyond the scope of this article.

The performance of the offsetting market depends on two main parameters: (1) the enforcement of offsetting liabilities that defines the offset demand (type of credits and quantity); and (2) the rules of the biodiversity banking system that determine the supply of offset credits (mainly the definition of ecological and geographical equivalence criteria setting the limits of the service area and the degree of asset specificity required). These two parameters actually depend on the institutional and legal contexts behind the BO device. The performance of the banking mechanism is, in fact, highly dependent on institutional and political choices.

From an empirical perspective, following the reports highlighting the inefficiency of the BO system in the USA, in 2008 the government set up the Final Rule to "improve the quality and success of compensatory mitigation projects" [51]. The definition of the 2008 Final Rule greatly reinforced the legal liabilities connected with BO by defining a specific framework, improving the control and monitoring of offsets, and providing precise rules (e.g., definition of a standardized method to assess ecological equivalence, requirement for funds for long term management, setting up a conservation easement, better organization of the banking system, *etc.*) [64]. Through the 2008 Final Rule, the US government encouraged developers to use the banking mechanism to conduct their offsets due to the banks' ecological and economic efficiency. The strong decrease in transaction costs to developers allowed through the banking mechanism and the reinforcement of regulations gave developers a stronger incentive to use this mechanism to meet their offsetting liabilities.

3.3. Offsetting Funds

In the offsetting funds system, also called "in-lieu fee mitigation" under the US legislation, developers pay a fee to specific entities, which differ depending on regulations (government, public agencies, non-governmental organizations, municipalities or environmental organizations). This mechanism also involves a third party who collects money from the developers and takes financial and legal responsibility for the success of offsets [70]. In this mechanism, the link between financial transfers and ecological gains is less direct and clear. Studies of offsetting fund programs showed that these payments often result in poorly planned offsetting projects that do not provide sufficient ecological and geographical equivalences with impacts [71]. The performance criteria are less demanding than in the banking or direct offsets mechanisms (with regard to management funds, equivalence assessment methods, control and monitoring of offsets implementation) [72]. Moreover, offsetting fund programs often provide offsets well after impacts occur. Thus, in most cases, offsetting funds do not provide sufficient guarantees that NNL of biodiversity will be achieved.

Offsetting fund programs face a risk of underestimating the funds required to conduct and achieve offsetting projects, or even to failure to perform the expected ecological actions [73]. In addition, the unclear link between the impacts of projects and offsetting requirements may mean that developers fail to take biodiversity into account when planning development projects [7]. In practice, offsetting funds are rarely incorporated into regulatory frameworks; they are often accepted in exceptional cases or in addition to other offsetting measures [7].

4. Main Structural Limitations for the BO Approach in Meeting the Biodiversity Conservation Objectives

4.1. Ecological Limitations

- **Limitations in integrating ecological knowledge through BO practices**

A major limitation of the BO system is that it is primarily based on incomplete and imprecise scientific knowledge regarding biodiversity and conservation issues [17]. Most of the practice of BO implicitly rests on scientific knowledge in the fields of ecology, conservation biology and ecological restoration. These disciplines remain relatively young scientific areas and face strong uncertainties with regard both to understanding biodiversity and its dynamics, and to predicting how it will evolve in a changing world [74,75]. In this context, the ecological success of BO remains uncertain and hard to predict. Restoration actions in particular have yielded mixed results, revealing difficulties in the recovery of the whole targeted ecosystem, with substantial and unrecovered ecological losses [54,62].

In addition to imprecise scientific knowledge, the BO system is conceptually impaired by the inherent difficulty of applying the most recent advances in ecology and conservation biology. Indeed, due to technical and operational limitations (e.g., time and spatial constraints in conducting environmental impact assessments and designing appropriate offsets), BO is constrained in practice to taking partial account of ecological scientific knowledge. Actually, the BO process is hardly influenced by current progress in scientific ecology, and a major gap is likely to result from the continuous mismatch between recent ecological researches and how biodiversity is treated in the BO process [76,77]. For instance, BO tends to consider species and habitats as isolated and static features of the ecosystem. However, this approach ignores a decade of research in ecology that has demonstrated the importance of adopting an even more systemic approach to biodiversity (*i.e.*, accounting for biodiversity dynamics, ecological interactions and processes) to consider higher degrees of ecological complexity (we define here ecological complexity by the property of ecological systems to be structured by multiple links and ecological interactions, emergent processes and non-linear dynamics) (Figure 1). Biodiversity responses to disturbance, for example, often show non-linear and not instantaneous dynamics [78]. Besides, in adopting a temporally and spatially restricted view of biodiversity during the assessment of losses, BO ignores the biodiversity potentially present (so called “dark-diversity”) [79]. The BO process also hardly considers the dynamics of biodiversity that result from processes interacting at different spatial scales from local to global [80]. Moreover, extinction debts may also be expressed long after disturbances, especially for species with long generation times, posing a major challenge for biodiversity conservation and BO [81,82].

This complex view of biodiversity makes it clear that biodiversity cannot be reduced to some of its isolated components (e.g., species or habitat), functions or utility (e.g., by adopting the lens of ecosystem services).

Moreover, switching from a systemic approach of biodiversity in the BO process to more functional or services approaches involves strong reductions in considering ecological complexity and represents an incomplete and less accurate view of biodiversity (Figure 1). First, ecosystem services are based on ecological functions which are judged to be useful to humans [83]. Therefore, one ecosystem service can be provided by different ecological functions. A complex lattice of species’ interactions, functional traits and dynamics are involved in ecosystem productivity, with varying degrees of usefulness. Ecosystem services are therefore only partially related to certain ecological functions; but many ecological functions cannot be equated with ecosystem services and can even constitute disservices (e.g., pollination is a function equated with a service for fruit production but with a disservice if invasive species are pollinated). There is no need for biodiversity to provide various ecosystem services that can be actually accomplished by human technologies (e.g., construction of a water treatment plant to provide the service of wastewater purification, pollination by hand to provide the pollination service).

Second, adopting a functional approach gives an incomplete view of biodiversity, as various and different species and ecosystems processes are able to provide the same ecological function (e.g., habitat function, climate regulation function). Moreover, ecological functions heavily depend on the structure and properties of ecosystems and on their distribution in space and time, which influence ecological interactions and ecosystem resilience [84]. Major ecological studies have thus emphasized the importance of adopting an even broader approach to biodiversity dynamics, ecological interactions and processes, resulting in a more thorough description of biodiversity. However, the reduction in considering the ecological complexity through functional or services approaches does not reveal the difficulties in assessing ecosystem services or ecological functions of biodiversity. Indeed, this is not because taking ecological complexity account is less high than assessments of ecosystem services or functions are easier to conduct.

Overall, the BO system fails to integrate and account for the ecological complexity of biodiversity in practice. Consequently, the BO process actually has problems accommodating the extensive contributions from ecology, especially when assessing biodiversity losses and designing equivalent offsets. The risk is that these failures may lead to underestimating ecological impacts, with the resulting incomplete or poor definition of equivalent offsets. From that perspective, BO may result in net loss of biodiversity, being unlikely to achieve NNL of biodiversity [61].

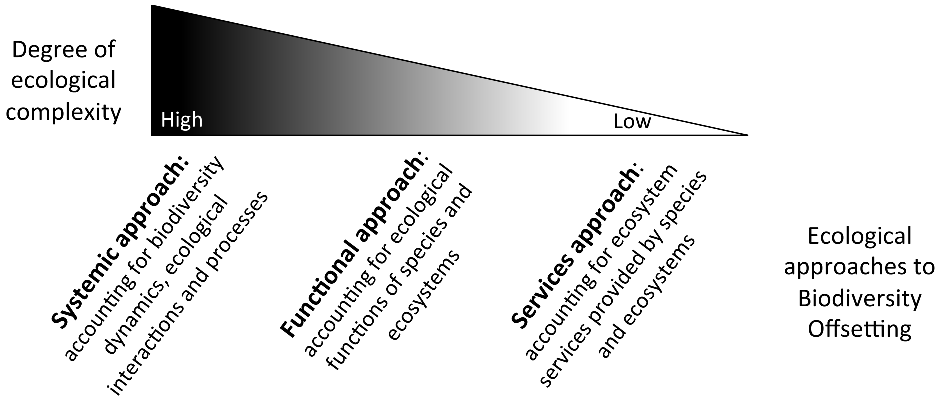

Figure 1. A complexity gradient across ecological approaches to biodiversity.

- **Limits to substitutability**

As Figure 1 illustrates, we argue that the degree of substitutability of biodiversity varies inversely with the degree of complexity of biodiversity (Figure 2). Mainly due to imprecise knowledge and technical difficulties both in accounting for biodiversity and in restoring ecosystems (as mentioned in the previous section), the higher the complexity of the biodiversity taken into account, the harder it is to reproduce the components of biodiversity, and thus to consider it as substitutable [85,86].

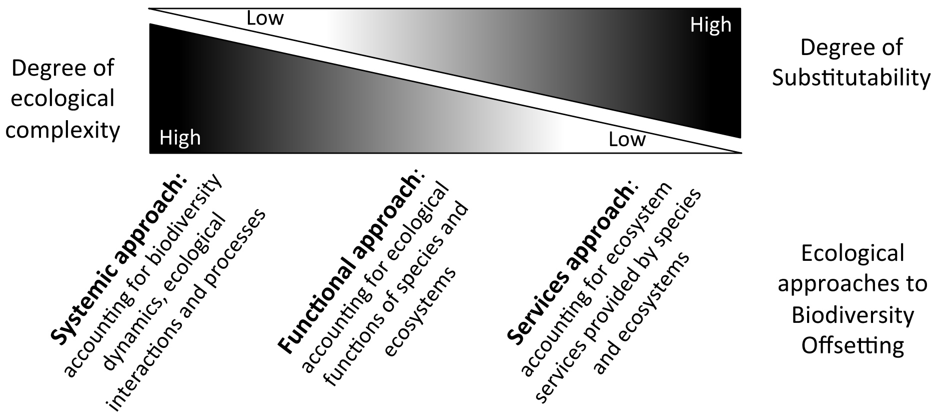

Figure 2. Trade-off between the degree of biodiversity complexity and its substitutability.

This leads to a paradoxical situation regarding BO. The NNL objective assumes a strong sustainability approach, which in turn implies low substitutability. However, the degree of substitutability depends on the ecological approach to biodiversity considered. It will be very difficult, perhaps impossible, to apply BO under a systemic approach to biodiversity because of impossibilities in accounting for all the biodiversity components. Thus, the NNL goal cannot be met easily when biodiversity is considered as being a highly complex object: the higher the degree of complexity involved in biodiversity losses, the less likely they are to be replaced and ecological equivalents found. And if we consider highly substitutable approaches of biodiversity *i.e.*, ecosystem services or functional approaches (Figure 2), we are failing to take the ecological complexity of biodiversity properly into account and this seems irrelevant from ecological perspective. This means that NNL policies and BO need to be defined and set up in such a way as to address the substitutability issues aiming at accounting the highest complexity gradient of biodiversity.

Placing the objectives of NNL policies within this context of substitutability constraints raises the question of NNL policy trade-offs. On the one hand, there is the will to implement an ambitious biodiversity conservation policy involving preserving a high degree of complexity in biodiversity, and on the other hand, BO risks falling short of the targeted results because of substitutability issues. In this situation, what kind of biodiversity should be targeted in NNL policies, and what are the most relevant ecological approaches to achieve it?

Answering these questions implies political choices based on the science available, and should, in our opinion, reflect a democratic choice of the kind of biodiversity we as a society agree to sacrifice and of the kind of biodiversity we decide to keep intact instead of pretending that win-win strategy can in most cases be found and scientifically supported. Indeed, the substitutability issue must be bounded by ethical concerns, based on social choice, for what we can offset and what we must preserve [87–89].

We suggest that the BO scheme be reserved for easily reproducible biodiversity *i.e.*, for ordinary biodiversity that includes many ecological equivalencies and allows considering simpler approaches of biodiversity due to lower conservation issues. The BO mechanisms cannot stand alone as a way to protect nature, but need to be backed up by properly-enforced public policy able to preserve non-substitutable ecosystems or species. Indeed, preventing the loss of biodiversity that we consider important to preserve from damage (such as endangered species and habitats) by strict statutory prohibitions remains the best way to guarantee no net loss of biodiversity.

4.2. Economic and Organizational Limitations

- **The risk of economic objectives prevailing over ecological objectives**

Through the use of economic incentives to preserve biodiversity, a new economic sector is emerging, featuring stakeholders new to the world of biodiversity conservation. This new economic sector involves, for example, environmental consultants, ecological engineering firms, companies and collaborative organizations whose primary aims are not necessarily those of biodiversity conservation. In the US where the offsetting sector is operating for a long time, the size of the overall annual market connected with BO is about USD 2.4–4.0 billion [8]. The BO system thus obviously encompasses more than purely environmental objectives; substantial lobbying now surrounds this market, with explicitly financial goals to be reached through "business solutions for a sustainable world" (http://www.wbcsd.org/home.aspx). The risk here is drifting away from a system aimed at preserving biodiversity towards a system aimed at ensuring economic outcomes [60]. This could have the perverse effect, as recent studies have shown [53,60], of encouraging some stakeholders to favor economic objectives over ecological objectives in order to preserve the link between the economic sector and the BO system.

- **The limited ability of economic design to meet ecological concerns**

While the banking mechanism is currently seen as the best way to perform BO from an ecological and an economic standpoint, this ecological and organizational efficiency carries risks. It could well encourage BO to be selected in preference to making initial efforts to avoid or minimize impacts on biodiversity. It needs to be remembered that the BO system necessarily implies ecological damage. Moreover, from an economic perspective, the proper functioning of this system implies assumptions on biodiversity assets. The banking system requires the most homogeneous and standardized biodiversity units to encourage trading of biodiversity credits. The more complex and specific the biodiversity credits, the harder it is for the offsetting bank to find buyers and sell its credits. In addition to the substitutability issues mentioned above, the economic mechanism behind the banking mechanism makes it difficult to target complex biodiversity with strong asset specificity. Even in the most effective mechanism, the banking system can thus lead to strong reduction in the complexity of biodiversity due to economic constraints imposing simplified biodiversity credits.

- **Organizational limitations: institutional risks**

In examining the BO mechanisms, we emphasized the importance of the institutional environment in ensuring good performance. The history of US legislation on BO shows that imprecise rules and the instability of the BO system leads in practice to offsetting failures. In fact, when an offsetting system is poorly designed and supervised, opportunistic behaviors can even lead to biodiversity losses. Nonetheless, some studies indicate that a certain flexibility needs to be maintained, for instance so as to allow the system to adapt to unexpected events resulting from environmental factors (e.g., species change in the context of climate changes) [90]. However, this institutionalization of the BO system depends on political will. Thus, under US and Australian legislation, the offsetting banking system is now well framed and regulated, leading to an improved BO system [49,50]. Better definition of the rules of the banking system encouraged bank operators to invest in conservation actions and developers to use the system. Elsewhere, especially in Europe, environmental regulations tied to BO liabilities have recently been significantly reinforced, but the banking system is still in experimental stages in most countries (France, United Kingdom, Germany) [8]. While this mechanism has begun to be introduced into environmental legislation, the design of current policy does not meet expectations because there are still many remaining institutional and organizational challenges to BO success.

5. Conclusions

The first objective of this paper was to clarify the economic background to the BO approach to biodiversity conservation. We showed that welfare economics and ecological economics offer relevant frameworks to analyze the BO approach. Whilst the basic assumptions behind these two approaches differ mainly in how they consider the substitutability and sustainability issues regarding capital, they both reveal the attractiveness of the BO concept and justify its use to address environmental externalities. However, they do not consider the same equivalence criteria and therefore do not involve the same performance criteria for BO. Welfare economics looks at equivalence in terms of utility, whereas ecological economics requires an ecological equivalence. Moreover, we showed that depending on the NNL policy goals, the BO system may consider different components of biodiversity involving different ecological approaches to biodiversity and resulting in different performance criteria and metrics used to assess losses and gains of biodiversity.

The second objective of this paper was to provide performance analysis of the three different BO mechanisms and highlight the main structural limitations of the BO approach in meeting biodiversity conservation objectives. Focused on ecological dimension, the banking mechanism ensures greater ecological effectiveness of offsets than the direct approach. However, in terms of ecological and geographical equivalence, the direct offsets approach is better at taking specific ecological features into account. From an economic perspective, the banking mechanism is more efficient than the direct offsets approach, but the economic constraints behind this mechanism can lead to inappropriate biodiversity conservation outcomes. Thus, defining a specific institutional framework and clarifying the regulations surrounding BO would appear to be crucial to the proper functioning of the system and the limitation of potential perverse risks.

Finally, we showed how the ecological limitations of BO point to a need to rethink NNL policies and BO goals in relation with the objectives of biodiversity conservation. In this sense, this paper offers a framework for debate on the balance between political will and ecological opportunities. Political choices are central to NNL policies, but they need to be based on the science currently available. In the light of ecological constraints, these choices will also necessarily involve public consultation.

Clearly, one of the main ways to improve the BO system is to better incorporate scientific contributions and social representation of biodiversity into the BO process. Ongoing ecological studies need to be used to support the increasing recourse to BO. Likewise, the scientific community should continue to investigate BO both in the natural and in the social sciences, providing both ecological and economics insights. While we have revealed the importance of economic factors within the BO process, careful investigation of how they operate in each specific project is needed.

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Author Contributions

Coralie Calvet, Claude Napoleone and Jean-Michel Salles conceived and designed the study; Coralie Calvet performed the research and wrote the paper. Claude Napoleone and Jean-Michel Salles also contributed to writing the paper. All three authors have read and approved the final manuscript.

Conflicts of Interest

The authors declare no conflict of interest.

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Article

Local Knowledge of Pond Fish-Farming Ecosystem Services: Management Implications of Stakeholders' Perceptions in Three Different Contexts (Brazil, France and Indonesia)

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Abstract: This article addresses ecosystem service perceptions in the case of pond fish-farming systems in Brazil, France and Indonesia. The Millennium Ecosystem Assessment vision suggests a more integrated reflection on environmental policies with greater adaptability to local knowledge and the development of social learning processes, which tend to promote more sustainable changes in behavior and practice than do sanctions. This study considers a part of the identification of ecosystem services. It shows that perceptions differ with the context, and found few differences depending on the type of stakeholders (fish farmers and other stakeholders). From a methodological viewpoint, this paper opens up new prospects for valuing ecosystem services through a perception study.

Keywords: social perception; ecosystem services; valuation method; pond fish-farming systems

1. Introduction

The Millennium Ecosystem Assessment [1] proposes a structural change in the reference framework for environmental policies, stressing the importance of reconciling the natural environment and human activities [1]. The ecosystem service approach includes several areas for research, such as identification, spatialization, monetarization, privatization and marketization of services [2,3]. Compared with the other areas, few studies specifically address the identification area. Yet it is generally accepted that the characterization of the services produced and used provides an operational framework for public policies [4].

In contrast, the monetary evaluation of services is very frequently undertaken, both to contribute to decision-making through cost-benefit studies or, more generally, to compare the weight and magnitude of such services. However, despite some improvement in the methods, this monetary approach continues to attract criticism. The main issues raised are not specific to ecosystem services and concern the assumptions about agents' preferences as well as measurement difficulties. For instance, Wegner and Pascual [5] emphasize the range of welfare economic dimensions that cannot be addressed though utilitarian and welfarist approaches. These authors note that "the results from environmental psychology confirm that ecosystems have relevance to human well-being far beyond the satisfaction of preferences including a strong bearing on psychological health, social integration and socio-cultural identity". Likewise, Hein *et al.* [6] highlight several constraints due to the range of viewpoints, depending on stakeholders and scales, together with double-counting risks [7]. They argue that it would be useful to categorize ecosystem services into intermediate services, final services and benefits in line with their economic definitions of services [8].

More generally, an analysis of the importance of these services and an assessment of their value has to build on prior knowledge. This is especially the case when, as stressed by TEEB [3], there is "no feeling of common heritage or legacy". Such knowledge varies with the actors and the areas. It can be addressed through the individuals' perceptions of these services [9]. These perceptions, which are central to risk analysis, enable the importance and the uses of these services to be identified and, through various methods, provide crucial input to public policies. They are particularly useful for understanding the acceptance of these policies [7,10] and, more importantly, for identifying potential voluntary agreements.

This vision suggests a more integrated reflection on environmental policies with greater adaptability to local knowledge and the development of social learning processes, which tend to promote more sustainable changes in behavior and practice than do sanctions [1]. These latter generate diversion strategies, whereas adaptability and social learning lead to appropriation. Such appropriation concerns two levels: (i) firstly, the appropriation of the ecosystem services by giving them value, not only use or exchange values, but also intrinsic value; and (ii) secondly, the appropriation of ecosystem management rules. Resource management is usually left to market forces, but ecosystem services lack the characteristics necessary for efficient market allocation, as they are non-excludable, non-rival, and damaged by negative externalities [11]. This change of reference framework increases the importance of more cognitive aspects to adapt public policies and promotes social learning.

We, therefore, studied these aspects through social perceptions. At the individual level, social perceptions determine the comprehension of behavioral development when facing regulatory measures to maintain ecosystems [10,11]. At the collective level, they determine the support for, and confidence in, the institutional mechanisms for the implementation of such measures. In addition, perceptions capture the degree of knowledge that the actors have of ecosystem services. Local ecological knowledge is an important element in the design and structure of natural management strategies [10,11].

This article addresses ecosystem service perceptions in the case of pond fish-farming systems in Brazil, France and Indonesia. The main goal of our research is the identification of the farmer and other stakeholder social perceptions in order to identify their knowledge of their working environment and the extent to which their perceptions correspond to the main measures of ecosystem management. This study considers part of the identification phase of ecosystem services. We worked in three countries with similar pond production systems but varying ecosystem and regulatory situations. In the first part, we outline the interest in studying perceptions and the methods used. The second part presents the cases studied. The third part presents the methodology we used to study social perceptions. The fourth part presents the results, which are then discussed in the fifth and final part.

2. Literature Review: The Interest in Studying Perceptions

2.1. Link between Perceptions and Environment

Social perceptions are organized and prioritized sets of judgments, attitudes and information of a given social group on a given topic [12]. Seca [13] argues that "*there are socio-cognitive and behavioural programmes affecting groups and their members*". The study of perceptions originated in sociology and psychology. Their application to environmental issues led to the development of environmental psychology [14] and its various currents depending on which factors are highlighted. They may be reduced to the so-called psychometric approaches [15], which focus on individual factors, and the culturalist approaches [16], which stress the role of social values and perceptions [17]. In practice, analyses usually include both types of factor. These aspects are at the heart of the economics of conventions. Understanding actor perceptions clarifies if collective conventions to which they refer and facilitates the development and implementation of coordination mechanisms exist. In particular, avoidance behavior with respect to norms and control measures can be reduced [18]. Livet and Thévenot [19] underline the role of collective conventions to make individual actions converged. Beuret [20] suggests that "*more flexible regulation on a case by case basis would be more effective than rigid poorly complied-with rules*". These conventions may be considered as "*collective cognitive arrangements*" [21] or as "*a set of collective behavioural rules*" [22]. This approach emphasizes institutional change processes [23], especially concerning meta-standard changes [24]. Such changes must be progressive and be accompanied by gradual implementation processes. This is the same principle of continuous improvement that is found at the core of the sustainable development framework and in the recommendations of the sociology of innovation concerning the adoption of new benchmarks [25].

2.2. Relevance of Perceptions in Environmental Policies

Although it has been little studied, the identification of the services provided is of strategic significance in the implementation of environmental public policies. Ultimately, the existence of an ecosystem service depends on the existence of a demand or a use (direct or indirect) or else on the recognition of a value (option or existence value). It is this demand or use that determines the contribution to social welfare. According to Bussard *et al.* (1998), cited by Schneiders *et al.* [26], *"Ecosystem management is managing areas at various scale in such a way that ecological services and biological resources are restored and conserved".* Balmford *et al.* [27] argue that the reasons and the values that underpin the interest of ecosystem preservation for human societies can be understood if perceptions are taken into account. These reasons and values allow farmers to become "*both production and ecosystem managers*" (Tilman *et al.*, cited by Dale and Polasky [28]). The identification of perceptions contributes to an improvement in the adaptation of measures and incentives that are put into place and hence in their increased acceptance by the actors [29]. The identification of services thought to be important by actors is a step towards understanding actors' perceptions and therefore their values. This means, in particular, identifying (i) particular target groups requiring specific accompanying measures; and (ii) the level of knowledge of services in order to define awareness-raising actions. Such actions should not be restricted to information, as not only the practices but also the values underlying these practices must be changed. Appropriation of environmental values favoring the recognition and conservation of ecosystem services requires specific organizational and institutional learning, so-called double-loop learning [30].

Numerous studies in environmental psychology have attempted to define the factors leading to the pro-environmental attitudes of environment conservation and protection and for consumers [31]. Among the most commonly studied are individual characteristics such as age, gender and education. Controversies surround the influence of age and gender but it seems that education and information have a consistent positive impact. This includes not only formal education but also tacit knowledge acquired through familiarity and proximity with the environment depending on use [18]. Several authors [32–35] show that actors' motivations and perceptions towards environmental conservation involve values of surpassing oneself. Those who are most sensitive to environmental conservation are often defined as altruistic compared to others who are more preoccupied with their short-term well-being and are seen as selfish. Most studies use the Schwarz value survey. Other collective factors such as social networks, and the standards and nature of management arrangements are also involved. Giddens [36] notes the importance of governance arrangements and more broadly of the trust in public agencies as a central condition for public policies to be accepted, which explains the importance of governance research [33,37]. The diagram below (Figure 1) shows our analytical structure. It summarizes the main components of the interactions between ecosystem services and socio-economic systems, showing in particular that users' perceptions are determined by both individual (social) characteristics and collective (public policy) characteristics.

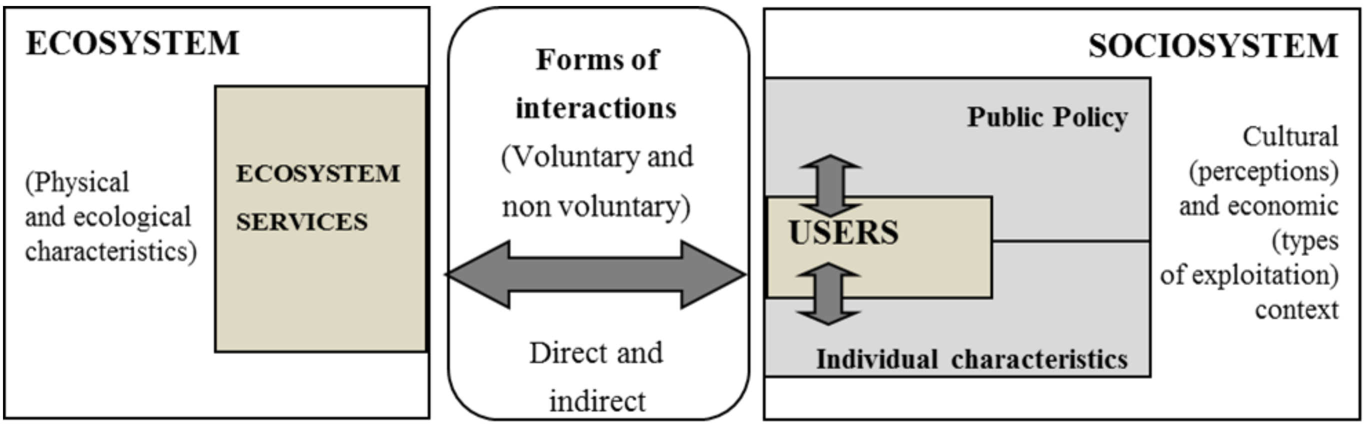

Figure 1. Analytical structure.

2.3. Need to Study Perception Diversity

Regardless of the determinants of behavior, analyses of perception tend to show that the range of uses and values depends on the context [38,39] implying that knowledge, social recognition and even social demand with respect to ecosystem services vary according to both the type of actor and the area concerned [40]. In order to be adapted, policies must therefore be defined and implemented according not only to the degree of anthropization (as noted by Schneiders *et al.* [26]) but also to the scales and contexts [6]. Prior to the monetary evaluation, some studies suggest qualitative or deliberative approaches (Sagoff, 2004 cited by Chan *et al.* [41]). These are particularly recommended when service uses are little developed and specific to a limited area [7] or when they depend on complex processes as with cultural services [41]. Chan *et al.* [41] decry the small number of alternative methods. They mention the multi-metric approach most often based on focus groups and collective evaluations. This approach uses "*subjective scaling when necessary*" [41] *or "ordinal ranking or numeric tag"*, or the creation of an index from 1 to 5 or 1 to 10. According to Chan *et al.* [41], "*many such constructed scales are in widespread use in society*. *Constructed indices can greatly facilitate a manager's decision by defining precisely the focus of attention and by permitting tradeoffs across different levels. Such a constructed index can focus a decision maker's attention on tradeoffs with other attributes and questions*". With that in mind, we designed a survey protocol to address perceptions of services and thus study their significance for actors and users without resorting to monetary evaluation. This type of approach converges partly on the issue of subjective indicators relating to people's satisfaction as a function of their perceptions about well-being. As Frey *et al.* [42] point out, these subjective approaches are less likely to lead to strategic responses. Nevertheless, this situated evaluation [43] depends on the context and means that all the levels at which ecosystems intervene and contribute to well-being must be taken into account.

3. Materials and Methods

3.1. Surveys in Highly-Contrasted Contexts

As part of a French research project (Piscenlit) funded by the National Research Agency, we identified ecosystem services provided by fish-farming ponds. The project seeks to identify ways to achieve ecological intensification [44] in fish-farming ponds to produce ecosystem services as both a pillar to sustain production and a means to diversify production. We studied three sites in Brazil, France and Indonesia (Figure 2):

- Recent multi-trophic systems based on recycling of effluents and utilization of byproducts with low nutritional value in Brazil (Santa Catarina State). Businesses are family-run and use earth ponds. The activity is associated with recycling farming effluents and the utilization of low-food-value inputs. Survey sites were selected in two areas of particular interest: the High Valley of Itajai in the East where the activity is highly structured, and that of Chapeco in the West where the organization is weaker, as it is in the rest of the region. These activities are situated in areas with large topographical variations. New laws have been implemented in order to professionalize the sector and reduce environmental impacts. They establish permanent preservation zones, "APP" (*Area de Preservacion Permanente*), and require installations to be at least 30 m from rivers. These measures seek to preserve the vegetation, the biodiversity and the functions of riverbanks. Training was provided for fish farmers in the Itajai area through an organization (Mavipi), within which all producers adhere to the same environmentally-friendly production method. The output goes either to the Sao Paulo market or for processing.

- Ancestral systems of extensive polyculture in France (Lorraine and Brenne) associated with recreational activities (angling, walking and nature observation). Brenne and Lorraine are key regions in pond aquaculture, representing 10% and 7%, respectively, of national production [45]. The production is mainly intended for enhancement (70% in Brenne and 90% in Lorraine). Many enterprises welcome visitors (fishing runs, fish sales on site, open days, and nature observation). These are situated in specific wetland areas subject to conservation measures (Natura 2000 or Ramsar sites). The activities in these areas are heavily regulated by environmental standards, especially the European Water Framework Directive. In Lorraine, agro-environmental measures have been implemented to strengthen the contribution of fishponds to environmental conservation. In Brenne, the ponds are part of a Regional Natural Park. Hence, in both regions, fishponds are key contributors to the nature of the landscape and the maintenance of biodiversity. Brenne, for example, is called the 1000-pond region. This feature contributes to its tourist attractiveness, particularly for nature tourism.
- Recent systems of semi-extensive monoculture in Indonesia (Tangkit and Kumpeh villages, Jambi Province). These activities are situated in areas focusing on pineapple crops. It is therefore a recent sector born of the conversion of agricultural holdings that have adopted traditional Javanese fish farming practices. These holdings are small and family-run (less than 1 ha in 98.5% of cases). Fish farming is carried out mainly in ponds dug for this purpose. Growth has recently slowed as a result of crises related to price instability and increasing feed cost due to pathologies, which are related to monoculture. In response to this crisis, which has led to a drop in the number of holdings, there has been a diversification of species and a growing awareness of environmental issues. These activities are heavily guided by the Minapolitan regulation, which aims to increase production by 353% by 2015. The objective of this program, managed by the Ministry of Marine Affairs and Fisheries (KKP), is to raise Indonesia, currently ranked fourth in global fish production, to first place.

Table 1 summarizes the characteristics of the pond fish-farming systems studied.

Table 1. Main characteristics pond fish-farming systems in the study area.

	Brazil	**France**	**Indonesia**
Surface Area (ha)	349 (Chapeco) 157 (Alto Vale do Itajaí)	7000 (Loraine) 8800 (Brenne)	292,000
Number of farms	932	242	576
Annual production (mt)	2373	2054	6935
Reared species	Tilapia, Common carp and Chinese carp	Common carp, roach, common rudd, tanch, pike, perch, pike perch	Catfish
Marketing	Local market and Sao Paulo market	Local restocking or fishing activities	National market
Specificities	Combined pig and pond system	Extensive polyculture; Many enterprises welcome visitors	Semi-extensive system with feed distribution
Productivity (mt/ha/year)	9.5–12	0.1–0. 4	89.9

Source: Brazil: Pro-mover and ADEMAVIPI [46], EPAGRI/CEDAP [47], France: FLAC [45], Indonesia: [48].

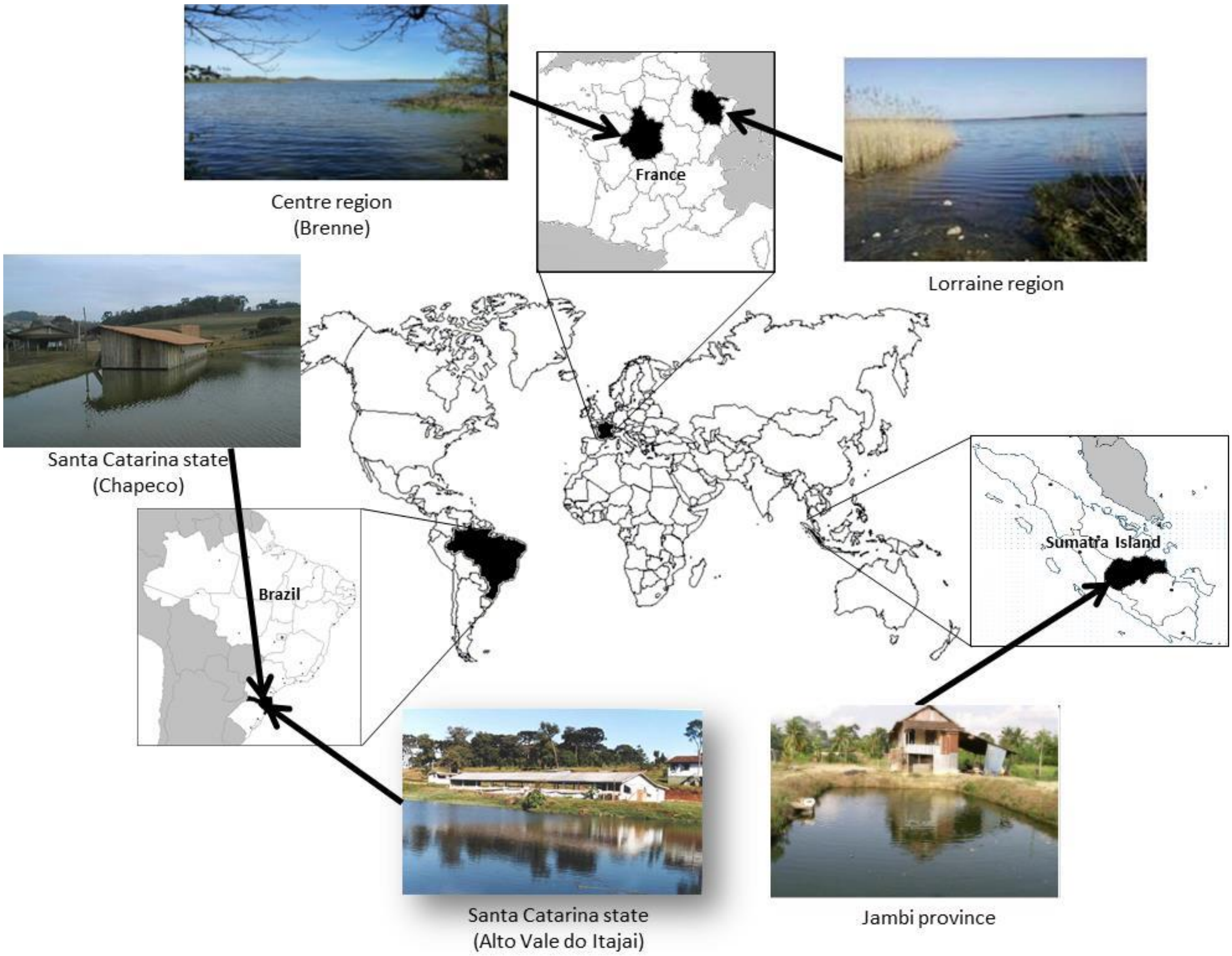

Figure 2. Map of the three study sites.

3.2. Methods

Few publications concern ecosystem services relating to pond farming. As no generic list specific to fish farming was available, we adapted the MEA list [49] to the case of pond farming in order to provide a reference list for the surveys. This adaptation was undertaken by the multidisciplinary group of researchers who are partners in the PISCEnLIT project using the literature and their knowledge [50]. The list was operationalized according to the specific context of each site. This operationalization was facilitated by many years' research experience at the study sites and by partnerships, in accordance with the recommendations and findings in the literature. This list contains 28 ecosystem services.

To assess local knowledge and ecosystem service perceptions, we interviewed farmers and stakeholders (Table 2) using semi-structured questionnaires that were developed on a multidisciplinary basis. Face-to-face surveys varied according to the context and lasted a couple of hours on average. In order to take diversity into account, the fish farmer sample was drawn from a stratified sampling frame.

Learning about perceptions requires adapted survey protocols [51–55], which purposely combine closed questions in order to establish typologies and open questions in order to analyze the spontaneous perceptions of interviewees. The questionnaire design took into account this recommendation. It combined open (spontaneous perceptions) and closed questions (ranking according to a pre-determined scale [56]). Following Kaplowitz and Hoehn [52], the open questions, placed at the beginning of the questionnaire, enabled perceptions to be identified without mentioning the notion of ecosystem services. The interviewees were then asked to rank the 10 services they valued most from a reference list. Given the large number of services and to avoid memorization issues, ranking was noted directly by the interviewee in the summary table. Unlike open questions, this list suggested services that may not spontaneously spring to mind. As well as the perception of services, the multiple structural and functional features of enterprises were also to be studied. In this paper, we analyze the responses to closed questions. A comparison between responses to open and closed questions is proposed in another paper.

Table 2. Structure of surveyed samples.

	Brazil		**France**		**Indonesia**	**Total**
	Chapeco	**Itajai**	**Lorraine**	**Brenne**		
Number of enterprises	690	242	42	200	576	1750
Diversity in enterprise type	Two types	One type only	Very high	One type only	Two types	-
Surveyed sample	50	25	25	33	34	167
Sampling ratio	7%	10%	59%	17%	6%	10%
Number of other stakeholders surveyed Of which:	34		59		9 *	102
State and administrative authorities	14		31		2	47
Associations and professional organisations	5		15		1	21
Downstream and upstream value chain	15		13		6	34

* 29 stakeholders interviewed but only 9 ranked the services.

Two indicators were calculated from the prioritized services:

- citation frequency, which represents the number of times each service was selected and therefore considered important; and
- an average score, which is the sum of scores obtained during the prioritization of those services that were considered to be the most important.

In order to identify the nature of the services most frequently mentioned, we used the citation frequencies, which turned out to be the most reliable data, regardless of country. In Brazil, as fish farmers found it difficult to rank the services, there was insufficient data in terms of score by service to be usable. Hence, in order to compare the three countries, we calculated for each service the percentage of producers and of other stakeholders who mentioned this service compared to the total number of producers and other stakeholders interviewed in each country.

Given the great diversity of services, we decided to analyze the results by classifying services using several macro-categories. Rather than use again the widely-used categories of direct and indirect services [7], we opted to use categories based on the value of the natural capital proposed by Petrosillo *et al.* [57]. Petrosillo *et al.* [57] distinguish nine categories, which we have grouped into three: (1) the economic value related to the economic opportunities generated by the ponds; (2) the biological value related to the supporting and regulating services; and (3) the cultural value ,which refers to heritage and recreational services.

Thus, in this paper, we allocated the listed ecosystem services to three value categories (economic (8 services), biological (8 services) and heritage (12 services)) and, using a comparison method, we analyzed the service selection percentage (citation frequencies related to number of producers or other stakeholders) mentioned for producers and other stakeholders. Summary tables showing all the results obtained by decreasing order of frequency are presented in the supplementary material section (Tables A1 and A2; the ten top selected services are highlighted in grey in the table).

4. Results

Given the significant number of services and our objective of comparing perceptions in three different countries, we decided to present the results in two ways, by service types and, generally (perceptions were compared between countries and between fish farmers and other stakeholders), according to the nature of their economic, biological and heritage value.

4.1. Results Per Nature of the Value of the Services

4.1.1. Comparing Perceptions within Services of Economic Value

Eight services (Figure 3) were found to present economic opportunities for producers. We found that Indonesian actors have the widest vision of this type of service. In Brazil and in France, the services selected by fish farmers and other stakeholders were fish production and the freshwater reservoir (the latter only for Brazil).

Fish production with its long history remains strongly present in perceptions despite the fact that it has become rather marginal. In France, and in particular in Lorraine, ponds were created by monasteries during the Middle Ages with the aim of developing fish consumption in non-coastal zones. This objective had much to do with the religious custom of eating fish on Good Friday. Until the French revolution, ponds were the domain of the wealthy classes and some 90% of them were the property of the nobility and the clergy [58]. This food supply function has greatly diminished and a large number of French pond farms are now producing fingerlings, either for restocking or for fishing activities with conservation or recreation in mind. Nowadays, pond conventions reward efforts made by fish farmers to maintain the environmental quality of their ponds. However, in anticipation of a fall in restocking demand due to the eventual banning of this practice in rivers, the industry is attempting to re-start consumption, in particular locally, drawing on the heritage character of this activity and on traditional recipes. In the case of France, despite the ancient origin of the activity, there are no services with a frequency over 50%, which explains the relatively weaker position than in other countries. However, it is in France that the diversity of heritage services mentioned above is the most important.

In Brazil and in Indonesia, the economic provisioning functions, mainly fish and freshwater but also fertilizer and plants in Indonesia, are the ponds' primary functions. These are developing countries where nutritional objectives are determinant, in particular in rural areas where poverty reduction policies emphasize the issue of food supply to local populations. Apart from this nutritional aspect, the differences between the countries may also be explained by the history of pond fish farming development and the nature of their integration in the areas.

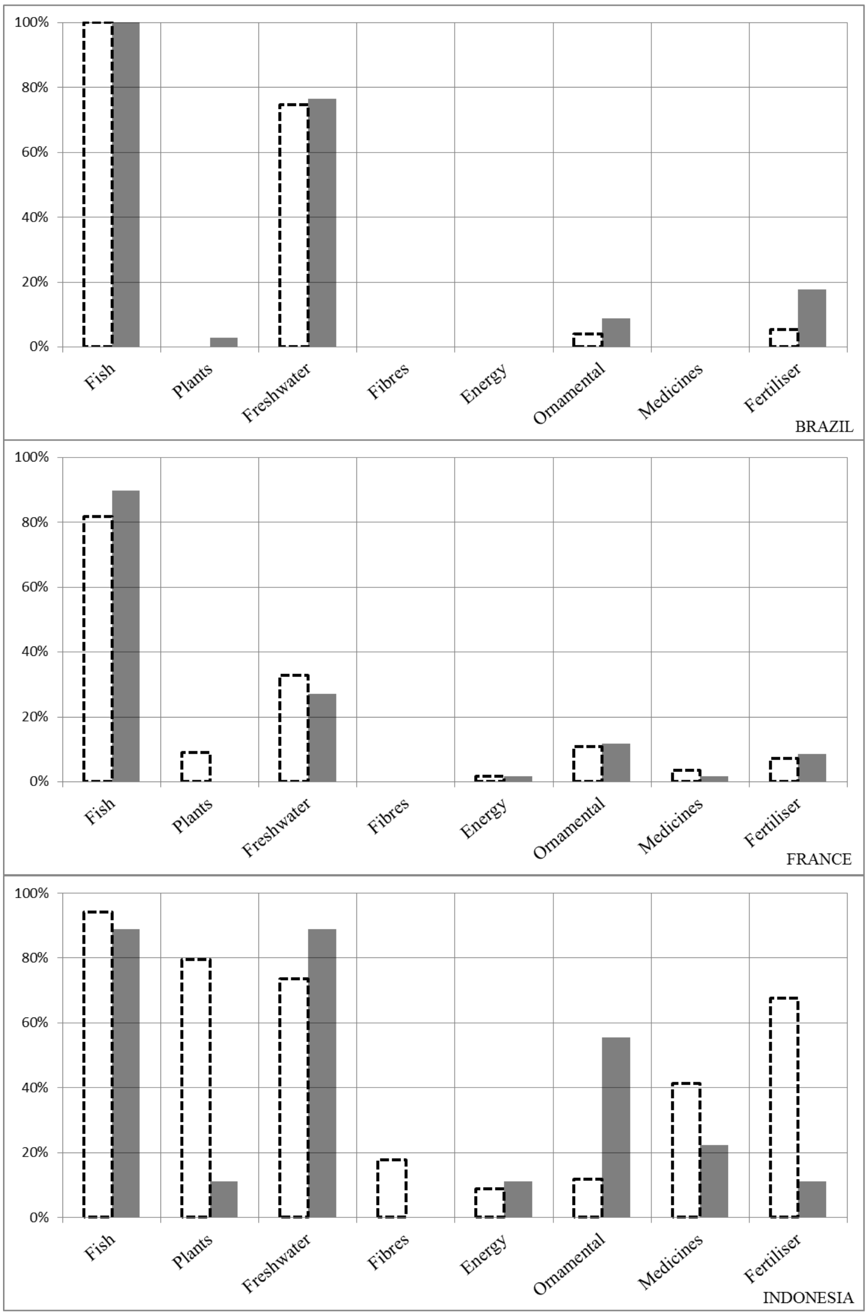

Figure 3. Fish farmers' and other stakeholders' selection percentage of services with economic value.

In Indonesia, it is services with economic value that were selected. Great importance is given to production. Since 2010, the Minapolitan law, which aims to increase production by 353% and raise productivity, has had a large impact on perceptions, as the Tangkit area is a pilot zone. As well as improved product quality, this law calls for "community" empowerment, which gives an important place to know-how. As in Brazil, the ponds are gradually generating leisure activities. These water bodies are tending to become walking areas for the local population and fishing activities are also developing.

Besides fish production, which is common to all countries, the diversity of supply services is greatest in Brazil and Indonesia, with freshwater reservoir and fertilizer supply, or plant production functions in Indonesia.

4.1.2. Comparing Perceptions within Services of Biological Value

Figure 4 shows that there was greater interest for services with biological value in Brazil and France. In Indonesia, only the other stakeholders select significantly a diversity of such services. Information relating to these services is much more widely disseminated in Brazil and in France, in particular through incentives that reward environmental services but also through restrictive measures arising from the legislation (e.g., FPA in Brazil or Water Framework Directive in France). In Indonesia, this type of information is somewhat less disseminated as incentives are more focused on increasing production, even if they include an environmental component.

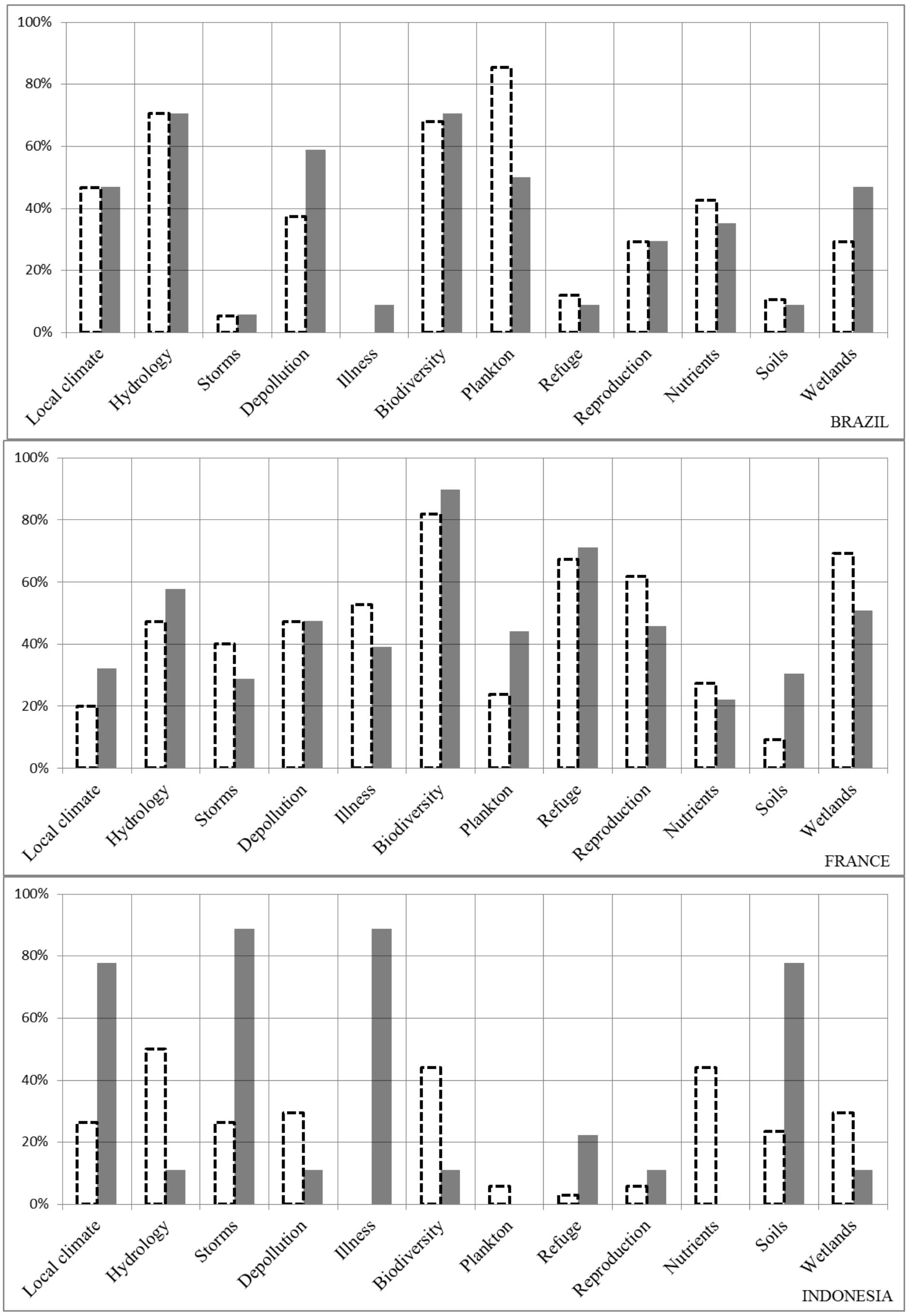

Figure 4. Fish farmers' and other stakeholders' selection percentage of services with biological value.

In the south of Brazil, production is not the primary activity because it is associated with the recycling of effluent from pig farming. Ponds were built at the instigation of the Federal State in the 1980s following a severe drought with the principal aim of creating freshwater reservoirs. Both Brazilian fish farmers and other stakeholders stress the role played by ponds during drought. Producers also insist on phytoplankton production related to the recycling function. However, both producers and other stakeholders highlight biodiversity protection. The actors are aware that the practice of recycling farming effluent puts heavy pressure on the environment, which they seek to reduce. The State has enacted a law creating permanent preservation areas and requiring installations to be at least 30 m from the rivers. Professional associations, in particular in Mavipi, advocate production methods that respect environmental constraints using agro-ecology principles [59].

In Indonesia, the importance of water regulation and storm protection can be explained by the buffer role of ponds in these areas, which are prone to flooding.

It is in the case of services of biological value that frequencies are the lowest. This may be because ponds are often built independently from watercourses and people's awareness of environmental consequences and of the advantages generated by these practices is of recent origin. Lastly, the profiles of the support and regulating services differ greatly depending on the importance and orientation of public environmental policies (existence of agro-environmental measures in France), type of farm (integrated pig-fish farms in Brazil that focus on phytoplankton production), and contexts (buffer role played by ponds during floods in flood-prone zones in Indonesia).

4.1.3. Comparing Perceptions within Services of Heritage Value

In Brazil and in France, the other stakeholders generally have a broader vision than fish farmers on services with a heritage value (Figure 5). In Indonesia, we again found a difference in viewpoint between fish farmers and other stakeholders for this category of service.

Likewise, the importance of landscape and recreation aspects, as well as leisure-fishing activities and hunting is explained both by lifestyle and the heritage character of ponds built in the Middle Ages. In Brazil, leisure and awareness-raising services are highlighted, due to their introduction as the principal activity of some enterprises along with a gastronomic element. In Indonesia, recreational aspects are recent and limited to fishing competitions. Also, the importance of know-how is explained by learning issues, which are related to the short history of the type of fish production studied.

4.2. Comparing Perceptions between Countries and between Fish Farmers and Other Stakeholders

4.2.1. Comparing Perceptions per Country

We observed large differences in perception linked to (i) the type of physical context, which influences the role of the pond (e.g., position in the watershed, size, number of ponds); (ii) the history and age of the fish-production activity; and (iii) the diversity of practices, uses, and public policies related to ponds. The table below (Table 3) summarizes the results according to the number and the importance of services mentioned in each country (number and percentage of services with a relative frequency over 50%). It shows a tension between the biological, and to a lesser extent the heritage, value and the economic value. The respective importance of these values can be related to the history and age of fish farming activity in each country.

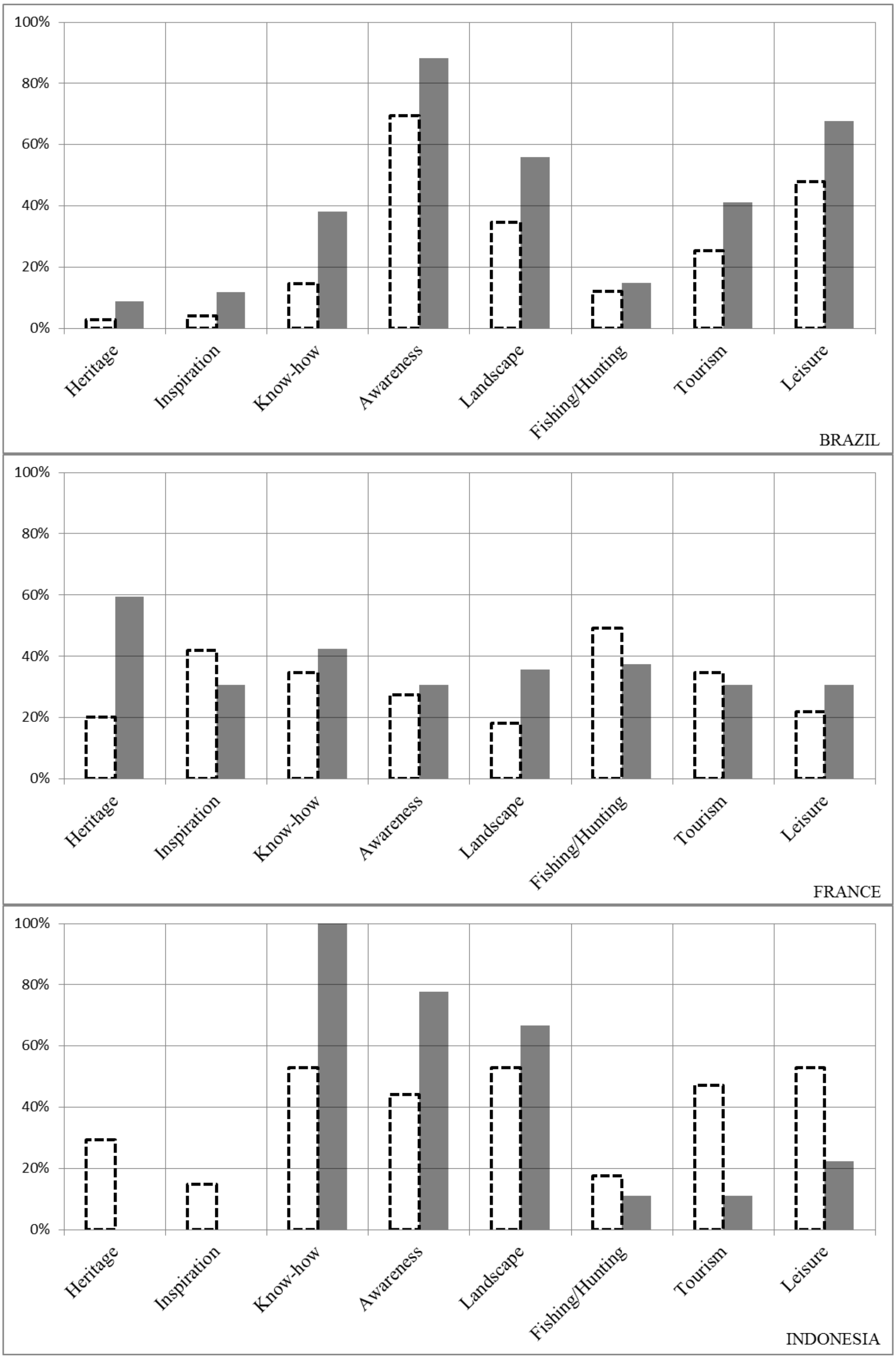

Figure 5. Fish farmers' and other stakeholders' selection percentage for services of heritage value.

Table 3. Summary of the relative importance of values as a function of the diversity and the citation frequency of ecosystem services (number and percentage of services with a relative frequency over 50%).

	Economic Value	Biological Value	Heritage Value
France	1 (13%)	5 (42%)	0 (0%)
Brazil	1 (13%)	3 (25%)	2 (25%)
Indonesia	4 (50%)	0 (0%)	3 (38%)

These differences may be explained first by the geographical characteristics of the areas and the forms of fish farming undertaken and second by the diversity of uses of, and public policies towards, the ponds. For example, pond areas in highly urbanized contexts such as France constitute "natural" and landscape areas, which are very attractive to both the local population and tourists, mainly ecotourism due to the presence of birds and nesting grounds in lagoons and wetlands. It should be noted that the hunting/fishing service (a heritage function in these areas) is only cited in France.

As shown by Hein *et al.* [6], in the case of the Dutch wetlands, the primary function is no longer the presence of anglers (fish farmers in our case) but the generation of significant cultural and heritage value. These values, related to environmental conservation, are strengthened by pro-conservationist public policies (Ramsar, Natura 2000, Nature Park). It should be noted that the service of biodiversity preservation has begun, in some regions, to be rewarded under agro-environmental measures. Fish farming has an important role to play here because in the absence of this activity there is a tendency to close lagoons to promote hunting, which is detrimental to biodiversity.

4.2.1. Comparing Perceptions per Country

Between types of actors (fish farmers and other stakeholders), differences in perception can be explained by differences in the scale of approach, levels and forms of knowledge (family-based or academic), as well as the degree to which the family is involved in fishponds. There are strong similarities in their viewpoints in France and Brazil, where information about ecosystem services is the subject of awareness programs or incentive measures. There are, however, a few differences: for example, the other stakeholders in France and Brazil have a wider vision than fish farmers about the heritage value of ponds. Likewise, in Brazil, the other stakeholders are more conscious of the importance of know-how and the part played by the landscape. In contrast, perceptions differ more in Indonesia, where fish farmers and the other stakeholders do not rank services in the same order. The degree of these differences is due to a certain institutional and cognitive "isolation" of fish farmers who are recently-converted farmers and thus have highly variable educational levels. The table below (Table 4) summarizes the differences observed as a function of the number of services where the frequency variation between fish farmers and other stakeholders is at least 15%. It shows very contrasting situations between France and Brazil on the one hand, and Indonesia on the other.

These differences in perceptions show the need to explore a wide diversity of viewpoints. Consideration of this multiplicity can help in understanding the diversity of position, which may restrict the acceptance of certain management measures.

Table 4. Summary of perception differences between fish farmers and other stakeholders (number and percentage of services where the frequency variation is higher than or equal to 15%).

	Economic Value	Biological Value	Heritage Value	Total
France	0 (0%)	2 (17%)	2 (25%)	4 (15%)
Brazil	0 (0%)	3 (25%)	3 (38%)	6 (23%)
Indonesia	4 (50%)	7 (58%)	4 (50%)	15 (58%)

Despite the differences observed between the countries, the comparison of perceptions between fish farmers and other stakeholders shows, in contrast with the literature, a large number of similarities for Brazil and France. These similarities may, perhaps, be explained by the indicators that we were forced to use, as citation frequency is a less subtle measurement of perceptions than service ranking scores. The scores tend to emphasize the differences [55,60].

5. Discussion

5.1. Comparing Perceptions

In line with the literature [6,7], our observations show that service perceptions differ with the context. Generally, such differences between fish farmers and other stakeholders can be explained by differences in scales or knowledge. As noted by Hein *et al.* [6] and Fisher *et al.* [7] "*Stakeholders at different spatial scales have different interests in ecosystem services*" [7]. Alongi (2002) also shows that local residents prefer provisioning services whilst wishing to maintain regulating services and that national and international actors are essentially preoccupied by the loss of mangroves and biodiversity. In our case, similarities are mainly due to very specific governance systems of the value chain in Brazil and in France. In the case of France and in particular in Lorraine, there are few fish farmers and they tend to be as highly educated as other stakeholders. But the most important factor is that there are close ties between industry and research organizations leading to the dissemination and appropriation of the standards underpinning public policies. In one Brazilian site, fish farmers have a similar close relationship, not with research but with a well-structured and very active association: ADEMAPIVI, which acts as an interface between research and government. This association is developing a specific MAPIVI operating model which integrates strong awareness of the environment and agro-ecological principles [59]. Precise practice guidelines and training for the industry are defined on this basis. By contrast, in Indonesia, the level of education is generally low in the industry, which tends to be somewhat isolated institutionally and cognitively. It should be noted that most producers are farmers who have only recently turned to fish farming. Indonesian fish farmers appear to have a broader vision of the economic opportunities provided by pond aquaculture. This is due to their risk management strategy, which is based on diversifying their activities. The production of ornamental fish was identified by the other stakeholders, but it requires a degree of technical expertise, concerning the selection of ornamental fishes for example, that goes beyond existing local knowledge.

5.2. Contribution to Decision-Making and Evaluation

Several authors stress the value of a holistic and integrated approach in improving interactions between human activities and environment. The ecosystem service approach provides a potential mode of action which can be guided by the market, the State (respectively "invisible" and "visible" hands) or by human values [61]. The latter approach, called "the third hand" by Wang *et al.* [62], may be complementary in the management of ecosystem services insofar as values affect preferences and have an impact on decision-making and individual and collective behavior. The same logic underpins the economics of convention which is why "*cultural values and social norms exert strong influences on and can dominate socio-economic policies*" [62]. Taking values into account through the analysis of perceptions thus represents a new approach in economics which leads to a questioning of the nature of the evaluation [63] to support the public decision-making process. Evaluation is indeed essential to assist decision-making in service management. Such evaluations can be monetary or non-monetary and both are important for decision-makers [64]. According to Daily *et al.* [64], there is a genuine need for non-monetary methods in service evaluations. The measurement of the value of services is multidimensional and varies by type of actor. Using non-monetary methods takes into account the diversity in viewpoints, the order of preferences [65], and the incommensurability of nature [66], and is in line with the adaptive and participatory governance of natural resources suggested by Ostrom [67]. This type of adaptive approach allows for the integration of changes in preferences that occur over time [68,69].

6. Conclusions

The aim of our analysis was to demonstrate the interest in studying perceptions in support of public policies. We hypothesize that an understanding of the services provides a positive signal towards the acceptance of policies for their conservation. One important element in the ecosystem approach is local knowledge supported by knowledge transferred by stakeholders, awareness-raising initiatives and the knowledge internalized by the actors [30]. Local knowledge is very important because it facilitates the understanding of complex context-specific ecosystem processes. The integration of this type of knowledge can help to develop production practices, for example towards agro-ecological benchmarks [59]. Commercial aquaculture has tended to lead to the loss of tacit know-how that may be the cornerstone in the implementation of ecosystem service management [70]. However, a certain degree of inertia must be taken into account, related, for instance, to the career paths of fish farmers, which emphasize the importance of knowledge transfer and of capacities for new learning [70].

We should highlight the fact that ecosystem management measures may be improved if they integrate locally-based information provided by farmers using social perceptions with global and empirical perspectives provided by scientific data. Local knowledge often provides guidelines and new information for ecosystem management and reciprocally strong regulatory measures influence perceptions. In fact, Silvano *et al.* [11] argue that "*a landowner with incomplete knowledge of the ecosystem services provided may therefore give them less weight than direct market benefits*". This observation argues for the involvement of farmers in the development of ecosystem management schemes. This involvement has to extend to the more general unit of stakeholders [65].

Our surveys, undertaken in three very different contexts (Brazil, France and Indonesia), have shown, in line with previous research, that perceptions differ with the context, and in particular in our case, with the history and age of the fish farm in the relevant territories. On the other hand, with the exception of Indonesia and apart from a few differences depending on the type of actor, perceptions were found to be fairly similar between fish farmers and other stakeholders. Compared to previous research, these similarities are novel and may be explained by the governance systems in place within the value chains.

This study of the perception of services was undertaken, service by service, in given sites. New and more global issues must also be addressed. Barbier [71] recommends a spatial approach to the distribution of these services. This type of approach provides an opportunity to reflect on the interactions between services and threshold effects in the conservation of a service. For example, the evaluation carried out by Barbier [71] of services relating to nursery grounds and the protection of mangroves and wetlands against storms showed that a minimum size was necessary if these two services were to be sustainable. These threshold effects require the recognition of ecological solidarity between territories [72]. They also require a change of scale in the implementation of conservation policies towards ecological corridors [49] that link protected areas. Furthermore, ethical issues must also be considered as they can play a role in the equitable distribution of these services as ecological amenities. This leads to the heavily-studied issue of ecological inequalities. Such spatialized approaches require the services of geographers and the integration of multilevel governance processes.

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Author Contributions

Syndhia Mathé and Hélène Rey-Valette were involved in the conception and design of the questionnaires. Syndhia Mathé analyzed the data. Both authors wrote the paper. Syndhia Mathé contributed to research materials and analysis tools. Hélène Rey-Valette contributed to the literature review and theoretical foundations. All authors gave thought to the discussions and conclusions parts. All authors read and approved the submitted manuscript.

Conflicts of Interest

The authors declare no conflict of interest.

Appendix

Table A1. Presentation of the services selected by fish farmers in each country (% of fish farmers having selected this particular service).

Brazil		France		Indonesia	
Fish/crustacean production	100%	Fish/crustacean production	82%	Fish/crustacean production	94%
Plankton production	85%	Maintenance of biodiversity	82%	Food plant production	79%
Freshwater reservoir	75%	Refuge and nesting	67%	Freshwater reservoir	74%
Water regulation	71%	Spawning and reproduction areas	62%	Supply of fertilizer for agriculture	68%
Raising environmental awareness	69%	Participation in natural nutrient cycles	62%	Learning of “know-how”	53%
Maintenance of biodiversity	68%	Hunting and fishing	49%	Landscape and attractiveness	53%
Leisure	48%	Pollution storage, depollution	47%	Leisure	53%
Local climate regulation	47%	Water regulation	47%	Water regulation	50%
Participation in natural nutrient cycles	43%	Source of inspiration	47%	Tourism/ecotourism	47%
Pollution storage, depollution	37%	Protection against storms and floods	40%	Maintenance of biodiversity; raising environmental awareness	44%
Pollution storage, depollution Landscape	37% 35%	Protection against storms and floods Learning of “know-how”, tourism	40% 35%	Participation in natural nutrient cycles Medical resources	44% 41%
Spawning and reproduction areas; Protection of wetlands	29%	Freshwater reservoir	33%	Protection of wetlands	29%
Spawning and reproduction areas; Protection of wetlands Tourism	29% 25%	Freshwater reservoir Raising environmental awareness	33% 27%	Pollution storage, depollution Heritage resources Protection against storms and floods	29% 26%
Tourism Learning of “know-how”	25% 15%	Participation in natural nutrient cycles Leisure	27% 22%	Local climate regulation Soil maintenance	26% 24%
Hunting/Fishing; Refuge and nesting	12%	Local climate regulation	20%	Fibre production	18%
Hunting/Fishing; Refuge and nesting Soil maintenance	12% 11%	Heritage resources Landscape	20% 18%	Hunting and fishing Inspiration	18% 15%
Protection against storms and floods; Fertilizer for agriculture	5%	Ornamental resources	11%	Ornamental resources	12%
Inspiration; Ornamental resources	4%	Plant production	9%	Energy production	9%
Inspiration; Ornamental resources Heritage resources	4% 3%	Soil maintenance Supply of fertilizer for agriculture	9% 7%	Energy production Plankton production. Spawning and reproduction areas	9% 6%
Heritage resources	3%	Medical resources Energy production	4% 2%	Refuge and nesting Refuge and nesting	3% 3%

Source: 2011 surveys Piscenlit project.

Table A2. Presentation of the services selected by other stakeholders in each country (% of other stakeholders having selected this service).

Brazil		**France**		**Indonesia**	
Fish/crustacean production	100%	Fish/crustacean production	90%	Learning of "know-how"	100%
Raising environmental awareness	88%	Maintenance of biodiversity	90%	Fish/crustacean production	89%
Freshwater reservoir	76%	Refuge and nesting	71%	Freshwater reservoir	89%
Water regulation	71%	Heritage resources	59%	Protection against storms and floods	89%
Maintenance of biodiversity	71%	Water regulation	58%	Disease regulation (human and fish)	89%
Leisure	68%	Protection of wetlands	51%	Raising environmental awareness	78%
Pollution storage, depollution	59%	Pollution storage, depollution	47%	Local climate regulation	78%
Landscape and attractiveness	56%	Spawning and reproduction areas	46%	Soil maintenance	78%
Plankton production	50%	Plankton production	44%	Landscape and attractiveness	67%
Local climate regulation; protection of wetlands	47%	Learning of "know-how"	42%	Ornamental resources	56%%
Tourism	41%	Disease regulation	39%	Medical resources; Leisure; Refuge and nesting	22%
Learning of "know-how"	38%	Hunting and fishing	37%	Energy production; plant production; fertilizer for agriculture; hunting and fishing; tourism; water regulation; maintenance of biodiversity; protection of wetlands; spawning and reproduction areas	11%
Participation in the natural nutrient cycles	35%	Landscape	36%		
Spawning and reproduction areas	29%	Local climate regulation	32%		
Supply of fertilizer for agriculture	18%	Tourism; leisure; soil maintenance; raising environmental awareness	31%		
Hunting and fishing	15%				
Inspiration	12%	Protection against storms and floods	29%		
Refuge and nesting; ornamental resources; heritage resources; soil maintenance; medical resources	9%	Freshwater reservoir	27%		
		Participation in the natural nutrient cycles	22%		
		Ornamental resources	12%		
		Supply of fertilizer for agriculture	8%		
Protection against storms and floods	6%	Energy production; medical resources	2%		

Source: 2011 surveys Piscenlit project.

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Review

Participatory Environmental Valuation: A Comparative Analysis of Four Case Studies

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Abstract: The valuation of multiple ecosystem services requires the design of valuation processes able to integrate different dimensions of value and to cope with complexity. Following the "value-articulating institution" framework, we note that three core problems arise: the cognitive, normative and composition problems. Combining valuation methods, such as contingent valuation and multicriteria analysis, with participatory and deliberative techniques is increasingly promoted as a means to address those fundamental problems. However, the quality and legitimacy of the valuation process then becomes dependent on how participation is framed. We note that numerous issues need to be taken into account, such as the roles assumed by participants, the differences in contribution among participants, the level of participatory impact and the level of democratization of the decision-making process. This paper proposes a detailed qualitative analysis of four case studies, each of them having implemented a specific valuation method in a participatory process. We analyze how those cases were handled in each of the dimensions considered and offer our conclusions about the added values and remaining challenges related to participatory environmental valuation.

Keywords: environmental valuation; ecosystem services; value-articulating institutions; participation; deliberation; decision-making

1. Introduction

Increasing human pressure on the environment emphasizes the need to make explicit why the environment matters and how it can be taken into account in public and private decision-making. Within academic research and in political agendas, this concern is currently framed as the need to value ecosystem services [1,2]. The ecosystem services metaphor is criticized for its reductionism, as it tends to neglect the complexity of the relationships and interdependencies within functioning ecosystems and between ecosystems and socioeconomic systems [3,4]. However, the simultaneous valuation of multiple ecosystem services puts the recommendations of previous research results into practice, according to which environmental valuation should take into account multiple dimensions of value, without reducing them into one single scale [5]. This valuation debate has been a core issue in ecological economics [6,7] and it derives from more fundamental criticism leveled by ecological economists upon the flawed behavioral models of the rational individual in neoclassical economics [8,9].

This paper is not focused upon the various definitions and interpretations of the ecosystem service concept, which remains problematic [10–12], but it focuses instead on recent advances and remaining core issues regarding methods for environmental valuation. We develop an institutional perspective on environmental valuation, following the "value-articulating institution" framework [13–16]. We also draw insights from the participatory literature [17–19], as well as from the "deliberative ecological economics" research agenda [8,9]. The common ground of those approaches is to insist on the need for developing public and stakeholder involvement in environmental valuation, on the basis of a shared institutionalist perspective, according to which choices made about the environment take into account both collective and individual preferences, which are socially constructed through norms, rules, conventions and institutions [9,14].

The concept of "value-articulating institutions" emphasizes that valuation methods are sets of rules framing the valuation process (e.g., who is involved, what is data, how stakeholders articulate their preferences, *etc.*). This framework allows us to define three basic issues occurring when valuing the environment: The cognitive, normative and composition problems. We view these issues as fundamental, which means that they can never be completely eliminated. However, different valuation methods, such as monetary valuation and multicriteria analysis (MCA) present major differences when confronted with those three problems. The value-articulating institution perspective highlights that combining methods, especially monetary valuation and MCA with participatory approaches, is a credible means to reduce these methods' shortcomings.

Recent advances in the participatory literature also highlight the importance of participation to address complex and dynamic environmental problems [17], as well as the need to combine different approaches to capture different value dimensions [19–22]. From a similar standpoint, the field of "deliberative ecological economics" advocates for deliberative valuations to improve the quality of sustainable decision-making because deliberative processes are assumed to be more legitimate, fair and democratic [8,9].

Known methods such as deliberative monetary valuation [23] and social multi-criteria evaluation [24] represent such attempts to improve environmental valuation by combining traditional methods with participatory features. However, numerous issues remain regarding the implications for ecological and economic sustainability of using participatory approaches and deliberative methods [14,15,25]. For the purpose of this paper, we focus on four main issues: The roles assumed by participants, the differences in contribution among participants, the level of participatory impact, and the level of democratization of the decision-making process. Those criteria were designed based on the literature with regard to the information available in the cases studies considered.

To foster learning from empirical evidence, this paper proposes a qualitative comparative survey based on these four case studies, in which four different types of valuation methods were implemented with different participatory features. We selected the following cases on the basis of a literature review: In case A, focus group sessions were implemented before contingent valuation (CV) surveys [25]; in case B, the elicitation of willingness to pay (WTP) was done in a citizen's jury context [26]; in case C, various participatory methods (interviews and group talks) were combined with a CBA phase and a deliberative MCA phase [27]; and in case D, a MCA was realized in a stakeholder's jury setting [28]. Through this qualitative survey, the considered case studies are each equally valued as original research experiments, each having its own advantages and limitations. The qualitative methodology we used allowed us to better understand the relationships between the valuation methodologies that were used, the particular logics of the appraisals and the choices made by the research teams, as well as the specific institutional contexts in which they took place.

Section 2 develops the analytical framework that was used, and Section 3 is dedicated to the case study analysis. Section 4 concludes on the added values and challenges related to participatory environmental valuation.

2. Analytical Framework

2.1. The Cognitive, Normative and Composition Problems and the Value-Articulating Institutions Perspective

Environmental valuation is about formulating a choice over the consumption or preservation of environmental resources and attributes. In that process, social, economic and biophysical values are tightly interrelated: Both the biophysical consequences of the decision or project under valuation and the social appreciation of those consequences matter. According to Vatn, three types of problems arise during an environmental valuation [14].

First, there is a "*cognitive*" or an informational type of problem related to the difficulties in observing and weighting environmental attributes because ecosystems are characterized by a "*functional invisibility*" [14] (p. 308). This complicates the communication process and the emergence of a mutual understanding over what exactly needs to be valued. The cognitive problem challenges the assumptions of neoclassical economic theory, in which individuals are assumed to have known preferences, and emphasizes instead that participants and decision-makers often have to build their preferences during the process through improving their understanding of the objects under valuation.

Second, environmental valuation triggers a "*normative*" problem: that of (in)*commensurability*. Commensurability assumes that biophysical, social and economic values can be reduced to a single scale, implying the strong comparability of values either in ordinal (weak commensurability) or cardinal (strong commensurability) terms [5]. Assuming commensurability (*i.e.*, strong comparability) implies that individuals involved in the valuation process (participants and analysts) are able to do this operation or believe that it can be done. However, various ethical and moral dimensions as well as commitments and judgments can preclude commensurability and challenge the reductionist perspective on valuation (which applies for both economic and biophysical metrics). Environmental valuation processes can be designed to support incommensurability if the absence of a common scale of measurement is recognized: incommensurability does not imply incomparability, but weak comparability remains useful [5]. Taking incommensurability seriously also implies the recognition that different values cannot always be traded off against each other, or that there are limits to such trade-offs. In that sense, during a valuation process, the normative problem is connected with assumptions regarding *compensability*: *i.e.*, the idea that a loss in one type of value can be compensated for by gains in other types of value [5,24].

Finally, the third issue is the "*composition*" problem. It has to do with the "*functional indivisibility*" [14] (p. 308) of ecosystems. Indeed, ecosystems are functioning systems and processes characterized by complex interdependencies, irreversibility and threshold effects. In a system, trying to value components separately from the whole to which they belong, or assuming that those components can be traded off against each other is not relevant. Yet valuing nevertheless requires bundling the object under valuation. There is therefore a conflict between the holistic character of the objects under valuation and the finite nature of environmental valuation processes. Thus, compromises must be made during the valuation process about the degree of ecological complexity included in the analysis. This compromise is reflected in the choice of the ecological indicators and in their design. Taking the composition problem seriously implies a form of reflection upon the way that the objects under valuation are framed and upon the choice of the subsequent ecological indicators.

Defining valuation methods as "value-articulating institutions" emphasizes that: (1) Different valuation methods imply different forms of participation: who participates, in what capacity (e.g., as a consumer or as a citizen) and how (e.g., in a written form, orally, individually or collectively *etc.*); (2) Valuation methods differ in terms of what is considered data (e.g., observed prices, price bids, biophysical units, weights, arguments *etc.*); (3) They differ also according to the ways in which data and values are treated and articulated (*i.e.*, how data are produced and weighted, aggregated or agreed upon during the process). As a result, "*the process frames the outcome*": Different types of valuation methods, such as contingent valuation (CV), cost-benefit analysis (CBA) and multi-criteria analysis (MCA), do not equally address the cognitive, normative and composition problems [15].

CV and CBA rely on core hypotheses of the standard neoclassical model, where the key unit of analysis is the individual, framed by the behavioral model of rational choice. Preference utilitarianism implies strong assumptions regarding not only "what a choice is" (individual calculation, trade-off) and "how choices are made" (preferences are already given, commensurability is not questioned) [29,30] but also "what the environmental characteristics are" (externalities or commodities often unaccounted for in real markets). Typically, those methods do not adequately address the three core problems involved in environmental valuation [15], but they remain commonly promoted, partly for theoretical and sociological reasons, and partly for more pragmatic beliefs. Indeed, neoclassical economics remain largely dominant today, which explains that despites their major failings, CV and CBA remain viewed as more "objective", "systematic" and "scientifically well-grounded" methods. Furthermore, it is increasingly assumed that decision-makers speak the language of money and that they demand assessments based upon efficiency, on the basis of an evaluation of benefits and costs [4].

MCA stems from system analytics and can also be framed as an optimization process. However, the major difference is that MCA can be based on very different assumptions regarding commensurability, compensability and aggregation [5,24]. MCA typically offers interesting possibilities to address the cognitive problem because it allows precise structuring of the valuation in an impact matrix incorporating criteria, policy alternatives and weights. It can also be designed to address the normative problem if weights are designed as coefficients of importance [5,30,31], which is the case in outranking methods. However, a greater emphasis is placed upon how the problem considered can be deconstructed (*i.e.*, how the favored alternatives may change according to variations in weighting), which may lead to difficulties in reaching a final decision. Furthermore, when the choice over a policy strategy involves several decision-makers, the process of aggregating or articulating weighting preferences is a delicate operation.

The value-articulating institution perspective emphasizes that participation can help to reduce the shortcomings of CV, CBA and MCA and to better address the cognitive, normative and composition problems in environmental valuation. This is what we aim to assess by comparing the selected case studies. However, as the following section underlines, participatory formats have their own limitations and some issues should be handled with great care.

2.2. Participatory Environmental Valuation: Advances and Challenges

Participatory methodologies can assume either a non-deliberative approach or a deliberative approach. Non-deliberative methods include surveys, polls, public comments, public information sessions and public hearings, while deliberative methods include focus groups, citizens' juries, consensus conferences, deliberative monetary valuation, social multicriteria evaluation, advisory committees and visioning workshops [19,32]. Deliberation implies that all participants are gathered in one place with the explicit purpose of debating and exchanging information, ideas and arguments about the problem considered, after which either a final decision is made or the process is repeated.

Fundamentally, the quality and legitimacy of the outcomes of participatory valuation processes are heavily dependent on the choices made regarding the participation setting [17]. Indeed, participation "*faces a world of choices*" [19] (p. 21); and those choices influence all of the key dimensions of value-articulating institutions (who is involved, on what premises, how data are produced, *etc.*) as well as the outcomes of the valuation studies and their relationships to formal decision-making structures. Based on Videira *et al.* [19] and Zografos and Howarth [9], we propose a set of criteria useful to analyze complex participatory valuation experiments. The criteria were adapted from the literature to the selected case studies. The goal was to focus the analysis on a narrowed-down set of basic criteria that would allow us to exploit all of the information available in our empirical material, while avoiding redundancies in the comparison.

The first criterion playing an important role in participatory processes is the roles assumed by the participants. Indeed, participants can be addressed as consumers, as citizens or as stakeholders. In this paper, we define a stakeholder as an actor having a specific personal or professional interest in the environmental issue considered, or acting as a representative of the collective interests of a formally constituted group [33]. This can include both representatives of the political authorities in charge of management and other groups of stakeholders. By contrast, citizens are members of the broader public and act as representatives of the general interest. The issue of the role assumed by the participants is related to the basic question of who should be involved in the valuation, depending on the objectives of the study, the type of method applied and the context. This issue is connected to the identification of all relevant participants, to the representativeness of the valuation towards the general population and *in fine* to knowing who gets a voice through the overall valuation and how.

The second criterion assesses the differences in contribution between the participants. In complex participatory exercises, those differences can be related to the different stages at which various participants or groups of participants are involved in the process (e.g., during early stages only for problem scoping, or for designing indicators and policy alternatives, or during the assessment and the decision-making process, or afterwards for monitoring, *etc.*). Differences in direct contributions of participants during the assessment and decision-making stages are also included. The point is not to maintain in all cases a norm of equity in contribution among all participants; maintaining differences can be justified by the logic of the process. However, it remains an essential dimension to take into account when assessing the fairness and legitimacy of participatory processes, and it has strong implications on the quality of their outcomes.

Drawing from Arnstein's ladder of participation [32], Videira *et al.* [19] define five levels of participatory impact, ranging from information through consultation, involvement and collaboration to self-determination. This constitutes our third criterion. The level of participatory impact reflects the degree to which participants can determine the end product of the process, and it is associated with the orientation of informational flows between participants (e.g., one-way or two-way flows). It determines the types of outcomes of the valuation exercise (*i.e.*, if the participants had an influence on decisions, or on the design of the alternatives, or if they were able to make the decision itself).

Finally, the last criterion is related to the level of democratization of the decision-making process. Indeed, deliberative valuation methods are based on the normative claims of deliberative democracy, which criticize technocratic power and the mechanisms associated with representative democracy, to advocate instead that the direct and active engagement of citizens, through debates and a reflection upon preferences, is at the core of the legitimacy of public decisions. Furthermore, the underlying logic is that a deliberative process forces participants to think in terms of the general interest, which is likely to insure a stronger consideration of ecological issues [9,34]. Therefore, insofar as deliberation is promoted as a means to improve environmental valuation and decision-making, this is an important dimension to take into account.

The assessment of participatory outcomes can include other social goals such as learning, the inclusion of public values and preferences in decision-making, the potential to foster trust in institutions and the reduction of conflict between stakeholders [19]. However, those criteria were either difficult to assess on the basis of the information available in the case studies considered or redundant with other parts of our analysis.

3. Comparative Analysis of the Case Studies

3.1. Presentation of Cases

Table 1 presents the four cases analyzed. The value-articulating institution framework is used to identify key steps of the valuation processes and to establish a comparative grid for analysis. Reading Table 1 in columns allows a comprehensive understanding of each case, while a second reading following the rows provides interesting comparative insights.

Table 1. Case studies comparison based on the value articulating institution framework.

Case Studies	Holmes *et al.* [25]—A	James and Blamey [26]—B	Messner *et al.* [27]—C	Proctor and Drechsler [28]—D
Contexts and objectives of the study	Objectives of the study: Ecosystem services are generally unaccounted for in decision-making causing ecosystem degradation. Assessing the economic efficiency of restoration projects: Identifying which restoration scale provides the greatest cost-benefit ratio. Environmental problem: Restoration of riparian areas along the Little Tennessee River (LTR), North Carolina. Methods: CV/CBA associated with focus groups	Objectives of the study: Incorporating community values into environmental decision-making; improve the robustness of WTP values in CV/CBA. Environmental problem: Management activities of national parks supervised by the National Parks and Wildlife Service in New South Wales, Australia. *N.B. This case is focused on methodological dimensions. No real decision-makers were involved and it was presented as fictional to the jury.* Methods: Deliberative monetary valuation, *i.e.*, CV implemented in citizen's jury context.	Objectives of the study: Uncertainty and ecological complexity, flaws of CBA, decision-making quality, competence and fairness, stakeholder implication. Environmental problem: Water allocation conflict between locations (up-stream/down-stream) and users in the Spree River watershed, Germany. Methods: Integrated Methodological Approach (IMA) combining a large participatory process, CBA (single-criterion assessment) and deliberative MCA as different steps of the same process. *N.B. Only the CBA was realized when the paper was published. The MCA is only described.*	Objectives of the study: Identifying and prioritizing between ecological, economic and social dimensions; deciding upon a suitable and sustainable management strategy for tourism and recreational activities. Environmental problem: Severe environmental problems, including water allocation issues, caused by the annual influx of tourists in the Goulburn Broken Catchment of Victoria, Australia. Methods: Social multicriteria evaluation *i.e.*, MCA implemented in citizen's jury setting.
Elements under valuation	Ecosystem Services: Habitat for fish (abundance of game fish), habitat for wildlife (in buffer zones), erosion control and water purification (clarity), recreational uses (allowable water uses), ecosystem integrity (index of naturalness); five restoration programs/scales considered (current, small streams, small streams +2 miles, +4 or +6 miles)	Five management activities: Fire management (number of parks with good fire management), weed control (area controlled per year), feral animal control (area controlled per year), maintenance of visitor facilities (proportion of well-maintained) and management of historic sites (number of well-protected).	Long term variations (50-year projections) of net economic benefits for fish farming; lake tourism; public water management and lake water treatment and for ecological indicators such as mean water availability for minimum flow; average water flow for Berlin and for Spreewald. Five alternative management options and two scenarios (one taking into account climate change) are considered.	Ecosystem Services (water quality and quantity, biodiversity, aesthetics); social and cultural (public access to sites, jobs, cultural heritage and education) and economic dimensions (costs and benefits). Indicators include quantitative, qualitative indexes (scale of value) and binary indexes (presence or absence). Five alternative management options are considered.

Table 1. *Cont.*

Case Studies	Holmes *et al.* [25]—A	James and Blamey [26]—B	Messner *et al.* [27]—C	Proctor and Drechsler [28]—D
Participatory settings	Two types of focus group sessions: With experts to characterize relationships between ecosystems and their services and selected indicators; and (four sessions) with citizens to design CV surveys and predict results. Ninety-six respondents (consumers) to CV survey and statistical adjustment to the regional population. Citizen/consumer premises.	Citizen's jury composed of 13 randomly selected jurors through phone surveys, following stratification rules to ensure representativeness of the regional population. Five witnesses with particular expertise in each management activity and two witnesses on general national park management. The jury met over three days (preparation, presentations and deliberation). Citizen/consumer premises.	Twenty interviews and "snow ball system" to identify all relevant stakeholders. Group talk with one stakeholder group (cross-state group) around climate modeling and policy strategies and individual discussions about the impact matrix (CBA step). Deliberative outranking MCA with all stakeholders. Stakeholder premises.	Workshops and questionnaires before the jury. Stakeholder's Jury composed of five natural resources managers. Four witnesses (local water authority, local ski resort, state natural resources management, member of local parliamentary council) and a judge (community psychologist) assisted the jury during one day. Stakeholder premises.
Data	Computerized CV surveys with photographs and maps and specific biding structure (dichotomous choice). Expressed WTP represent the benefits associated with each restoration scale, while costs are estimated on the basis of similar projects implemented in the region, through a cost-sharing program of the Natural Resources Conservation Service. Net benefits, associated with marginal changes in ecosystem services provision.	Deliberation among jury members and debates with witnesses. Debates and argumentation around current management practices, comparison of alternative management options and qualitative suggestions. Individual WTP understood as the maximum amount that citizens could be charged given the environmental improvement and the payment vehicle considered.	Interviews and group talks. Climate change modeling. Co-production of alternatives strategies and criteria with decision-makers, based on interviews, group talks and data availability. Calculation and ranking of economic and ecological criteria depending on the five alternatives and the two scenarios considered (CBA step). Individual preferences (weights) identified by interviews. Arguments during deliberative step. Stakeholder preferences, policy trade-offs, future uncertainties and consensual alternative.	Preliminary phase (workshops and questionnaires): Development of management options, criteria, impact matrix and preliminary rankings. Arguments and debates around witnesses' presentations. Identification and discussion of juror's preferences (weights). Use of probabilistic software (ProDecX) to screen policy alternatives, discuss weights and to reduce uncertainty/dissensions around weights. Sensitivity analysis posterior to the jury.

Table 1. *Cont.*

Case Studies	Holmes *et al.* [25]—A	James and Blamey [26]—B	Messner *et al.* [27]—C	Proctor and Drechsler [28]—D
Valuation processes	Documentation on the historical characteristics of the region. Documentation on the cost-sharing program for riparian restoration to determine average costs of restoration (and the minimum benefits necessary for economic feasibility). Two types of focus groups were used for CV surveys design and for ES indicators, respectively. Computerized CV. Statistical analysis. CBA. -Aggregation-	The jurors are confronted with two charges: Under the first charge, jurors had to reach a consensus over three different options of management activities at constant budget. Consensus over status quo was reached. Under the second charge the jury had to consider improving all management activities financed by the introduction of a tax on inhabitants. No consensus was reached regarding the amount of the tax. A voting procedure was applied to close the process. -Consensus and voting-	Historical documentation and interviews. Development of scenarios and alternatives. Climate change and future uncertainties modeling and discussion. Modification of policy alternatives. Calculation of impact matrix. Individual discussions over impact matrix and identification of stakeholder's preferences (weights). Deliberative MCA: Individual impact matrixes are presented and discussed. A consensus has to be reached over weighting, otherwise new alternatives are designed and the subsequent steps repeated. -Consensual weighting-	Preliminary phase. Discussion around the outcomes after first ranking process showing strong dissensions among jurors. Witnesses' interventions. Replacement of the ranking process by a proportional weighting. Redefinition of the ES and social criteria. Discussion of new outcomes and justification of the weights assigned by jurors. Choice of a policy strategy. Sensitivity analysis showing a higher level of consensus. -Ranking and proportional weighting-
Outcomes for decision-making	Annual economic benefits (median WTP) for each restoration scale. Full restoration has the highest benefit/cost ratio. Decision-makers know that the biggest public benefits are associated with full restoration, on the basis of the restoration program in place (average costs) and the demands of the population (WTP statements).	Insights about current management alternatives. Arguments and counter arguments regarding the introduction of a tax on inhabitants. Partial agreement on a certain tax level, with discussions over equity issues. Possibility of including the WTP results in a CBA, comparing the amount of money that would be collected by introducing the tax with an estimation of the costs implied by the new management strategy.	Ideally, the process is able to assess and evidence for a consensual alternative for decision-making, taking into account weighted economical and ecological dimensions as well as inequalities in the balance of power between stakeholders and global external futures changes (socioeconomic and climate) over a conflicting situation. However, only the results of the CBA step are discussed in the paper.	Exchange of arguments and elicitation of decision-makers' preferences through weighting. Confirmation (after criteria redefinition) of a management option. After the process, decision-makers have another conception of the specificities of the problem considered. The confirmed management option is based on a higher degree of consensus than before the process.

Table 1. *Cont.*

Case Studies	Holmes *et al.* [25]—A	James and Blamey [26]—B	Messner *et al.* [27]—C	Proctor and Drechsler [28]—D
Limits	Challenges in linking ecosystem science with social values; difficulties in communicating complex ecological issues. CV respondents had trouble understanding how ecosystems should be valued (as substitutes or complementary).	Numerous issues are discussed: compliance behaviours, equity between jurors' contributions, inconsistencies between citizen framing and individual WTP elicitation, WTP interpretation, introduction of the voting procedure, articulation of CBA results and representativeness.	The authors underline that the process does not fully meet the ideal claims on which it is based, regarding the participation debate. However, it improves the decision-making process in terms of competence and fairness. Other important limits concerning time spending and costs are mentioned.	The authors mostly highlight problems with the software used for the weighting process and for the presentation of the outcomes to the jurors. They underline the necessity to discuss in details criteria and impact matrix as well as the importance of the iterative nature of the process.

Although the cases present a great heterogeneity in their objectives and contexts, a first point is that all of them are related to ecosystem degradation and land-use conflicts between various social groups. In all cases, a complex environmental choice has to be made considering an intended improvement of ecological conditions and a local development perspective, involving economic and social costs and benefits. However, if all of the cases studied are viewed by their authors as empirical tests of specific environmental valuation methods aiming at answering scientific research goals, then cases A, C and D are more directly concerned with a real policy issue than case B. Case B was implemented as an experimental test on deliberative monetary valuation.

3.2. Addressing the Cognitive, Normative and Composition Problems

This section focuses on the cases as empirical attempts to improve environmental valuation processes, by implementing them in participatory settings. The results are summarized in Table 2.

Table 2. Comparative analysis of the cases regarding the cognitive, normative and composition problems.

	A	**B**	**C**	**D**
Cognitive	Low/P	High	High *	High *
Normative	NSA	Medium	Medium/P *	High *
Composition	Low	P	High *	High *

NSA indicates "No Sensible Attempt" to address the issue considered; P indicates that the method could have been designed to address the issue but that it was not the case in the study considered; * indicates that because of the combined nature of the process, it is hard to assess whether the treatment of the problem was due to the participatory setting.

3.2.1. The Cognitive

In case A, cognitive issues were addressed through two main avenues: First, citizen focus group sessions were conducted with the aim of improving the framing of the information included in CV surveys (for instance, participants were presented with a matrix showing the level of ES provision associated with the different restoration scales), and second, the authors argue that using computerized instruments helped ensuring a better understanding of the bidding structure. However, in CV, individual preferences are considered as already given. In case A, the participatory setting did not attempt to address the issue of preference construction: Most of the respondents to the CV surveys did not participate in the focus group sessions. This explains the lower capacity to address cognitive issues in case A. More generally, attempting to address the preference construction problem by combining CV and focus groups would be problematic in our view because the logic of statistical representativeness implied by CV and the narrower format of the focus group setting would come into conflict.

In contrast, in case B, cognitive issues were seriously taken into account because of the deliberative setting: The jury took two days to deliberate, which certainly helped with the elicitation and construction of preferences. Furthermore, the authors discuss in depth how information should be provided to the jurors. They conclude in support of unlimited access to witnesses and propose to rely on one additional neutral witness dedicated to helping the jury with informational issues.

In case C, several interviews and group talks were conducted for problem analysis, the identification of relevant stakeholders and the design of policy strategies, indicators and scenario development (Table 1). This long phase of participatory preparation was very interactive, which probably already induced cognitive effects, *i.e.*, changes in the way each stakeholder perceived the situation. Furthermore, the cognitive dimension was addressed both during the single-criterion and MCA steps. The single-criterion valuation is a matrix measuring the quantified impacts of economic and ecological criteria dependent on the policy alternatives considered (Table 1). The authors name it CBA because it calculates and presents the rankings of all policy strategies with respect to each single criterion. The information structure is close to that of MCA, but no overall ranking or weighting of the criteria takes place. The point is to elicit trade-offs between policy alternatives and to make them salient in the minds of participants. The authors plan to discuss the CBA results and to collect preliminary individual weightings through interviews. Afterwards, during the MCA step, individual impact matrixes would be presented and the objective of the deliberation would be to attain a consensual weighting. If no consensus is attained, the process should be repeated. We conclude that the procedure used in case C has a high capacity to address cognitive issues, both because of the CBA/MCA structuring and because of the deliberative nature of the process.

In case D, management options and criteria (Table 1) were developed and discussed during a group talk prior to the jury, and questionnaires were sent to agree on the global objectives of the study and to identify preliminary individual rankings. Cognitive issues were especially addressed during the subsequent deliberative MCA step. First, jurors' perception of the situation was confronted with witnesses' presentations, after which debates, exchanges of views and arguments took place. This certainly had great cognitive impacts. Second, the MCA framing allowed the structuring information around an impact matrix showing quantified (cardinally and ordinally) relationships between criteria and policy strategies. The software used allowed the jurors to be aware of each other's individual preferences for the ranking of criteria. Furthermore, the integration of MCA and deliberation allowed the jury to discuss the results of the first ranking, choosing to modify the criteria structure and the weighting process along the way. The higher degree of consensus attained at the end of the process over a policy strategy can be understood as a positive consequence of the preference construction process. Similarly to C, the quality of the treatment of cognitive issues appears strongly related to both the deliberative setting and the MCA structuring.

We can conclude that in cases B, C and D, the participatory settings greatly helped to address cognitive issues.

3.2.2. The Normative

CV postulates commensurability. By definition, dealing with the normative problem is problematic. Furthermore, WTP results are used in CBA, which implies compensability. Interestingly, however, the scenario of local taxes increase proposed to CV respondents (WTP amounts were asked in terms of an increase in local sales taxes for different levels of riparian restoration) aimed to ensure the credibility of the amounts bid, and this procedure probably helped avoiding the potential reluctance of participants to state their preferences in monetary units. We did not find any information concerning the occurrence or treatment of 0-bids and/or non-responses, and it remains unclear in the study whether the respondents would feel comfortable with the interpretation of their WTP statements (*i.e.*, the measure of the social welfare that the environmental improvement considered would create for them).

In deliberative monetary valuation, the normative problem is generally a core issue. On the one hand, the process of WTP elicitation implies commensurability, *i.e.*, narrowing-down preferences to the expression of a monetary number. On the other hand, deliberative monetary valuation allows participants to think openly about their choices. In case B, as the process was designed to open the possibility of using the results in CBA, compensability was also assumed. The normative problem arose during the second charge, when the jurors were asked how much they would be willing to pay (in terms of a local tax increase) to improve all types of management activities. As in case A, the proposed scenario probably improved the credibility of WTP statements and helped avoiding the potential reluctance of participants to express monetary values for environmental attributes. The authors report that some jurors had trouble understanding the notion of individual WTP. Indeed, some jurors tended to adopt a "*contribution model*", instead of the "*purchase model*" typically assumed in CV and CBA: They wished to know how much it would cost to improve all management activities to make a decision about the amount of the tax. Because the process allowed jurors to exchange views and arguments, both about the problem considered and about what was asked them (e.g., the meaning of WTP statements and their expected use), the deliberative setting helped to raise and resolve normative issues. Because the deliberative monetary valuation process allows a certain degree of reflection upon WTP elicitation, we assign this procedure a medium capacity to address the normative problem.

The normative problem has a different form in cases C and D compared with cases A and B because the economic dimension is not based on WTP statements. Case C postulates commensurability and high comparability: The measurements included in the impact matrix are cardinal numbers. However, the MCA method used (PROMETHEE (Preference Ranking Organisation METHod for Enrichment Evaluations)) implied non-compensability. One interesting feature of the method used in case C is that the method allowed adjusting the assumptions regarding the normative problem to the considerations of the stakeholders involved. For instance, the use of economic net benefits was suggested and agreed upon by the stakeholders involved during the preliminary phase. This explains why we conclude for a medium capacity to address the normative problem in case B, without excluding the possibility for the method to be designed differently.

The same outranking method (PROMETHEE) was used in case D, and the economic costs and benefits were measured on the basis of existing data, depending on each policy alternative. The costs were mainly in terms of the establishment of facilities, weed control, fencing, lost incomes and visitor fees, while the benefits included increased incomes of tourist operators and accommodation providers. Case D implied a weaker form of commensurability compared with case C: The impact matrix included cardinal quantification, binary indexes and ordinal indexes. The criteria (Table 1) were also designed by the jurors at the beginning of the process, during the preliminary phase and during the jury. The jurors also decided to modify the weighting procedure to give the same importance to the three broad categories of criteria (economic, social and ES). We assign this procedure a high capacity to address normative issues, both because of the MCA structuring and because of the deliberative nature of the process.

We can therefore conclude that for cases B, C and D, the deliberative setting had positive effects on normative issues, but for very different reasons. As we noted in case B, the normative problem related to WTP statements did not disappear, but participants had time to think collectively about what was requested of them. In cases C and D, the normative issues depended on both the MCA structuring and the deliberative setting: The latter allowed adjusting the assessment to the considerations of the participants.

3.2.3. The Composition Problem

In case A, the focus groups sessions conducted with experts helped to address the composition problem through the characterization of the relationships between ecosystems and their services. Five categories of ecosystem services and their indicators were adopted (Table 1). The study did not attempt to offer precise quantifications of the level of ecosystem services provision associated with each restoration scale. Instead, the CV respondents were confronted with broad categories (low, moderate, high). One of the main results of the study was that the issue of scale can be taken into account in CV and that the benefits associated with ecosystem services provision are "super-additive" (*i.e.*, there is a holistic effect associated with the restoration scale). However, we should note that the study selected specific positively interdependent ecosystem services. This implies that the potential trade-offs between restoration activities and the other services that river banks provide are not taken into account. The authors conclude that "*much remains to be done to improve methods for communicating complex ecological dynamics in the context of economic valuation studies*" (p. 29), which indicates that both the cognitive and composition problems remain problematic.

In contrast, the composition problem is not particularly discussed in case B: The study focuses on changing management practices without attempting to precisely measure the expected effects of those changes on ecosystems (see in Table 1). In that sense, the composition problem was partly avoided. However, witnesses were experts in particular management activities. A dialogue between jurors and witnesses regarding ecological complexity, interdependencies, *etc.* could thus have occurred, but the authors do not document this point.

The composition problem was seriously considered in case C, but through climate change modeling than through the design of the ecological criteria: The latter were selected because of their relevance for the stakeholders involved (Table 1). The modeling was realized by the research team after the first round of interviews and group talks. Before the CBA phase, a group talk was organized during which uncertainties and failures related to the specific scientific models used were discussed. Therefore, the composition problem was addressed more through the use of the modeling tool than because of the participatory approach. However, the latter certainly helped raise and resolve cognitive issues related to the integration of modeling.

Finally, the composition problem was addressed in case D through the definition and redefinition of the ecosystem services criteria during the process. The nine ecosystem services criteria identified at the start were merged into four because they were considered redundant by the jurors after witnesses' presentation and the first criteria ranking. In that sense, the composition problem was addressed because the MCA provided information structure and quantifications and because the deliberative context helped the jurors to have a better understanding of the ecological interdependencies entering into play.

Overall, participatory settings proved able to better address the composition problem in all of the cases considered. However, except in case D, the selection of ecological indicators often relied on the choices made by the research teams or on the involvement of experts, rather than on the preferences of the participants involved.

3.3. Participation and Decision-Making: Issues and Differences between the Cases Considered

Attempting to improve valuation methods by introducing participation implies that the quality and legitimacy of the outcomes become dependent on how participation is framed. We noted four important criteria: The role models assumed by participants; the differences in contribution among participants; the level of participatory impact, and the level of democratization of the decision-making process. Table 3 summarizes our analysis for the case studies considered.

Table 3. Comparative analysis of the cases regarding participation criteria.

	A	**B**	**C**	**D**
Role models assumed by participants	Consumers and Citizens	Citizens and Consumers	Stakeholders	Stakeholders
Differences in contribution	Medium	Low	Medium	Low
Level of participatory impact	Consultation	Consultation	Collaboration	Collaboration
Level of democratization of the decision-making process	Low	Low/NA	Medium	Low

NA indicates “Not applicable” to the case considered.

3.3.1. The Role Models Assumed by Participants

Considering the overall process, the role assumed by the participants in case A varied depending on the stages of the process: Participants were considered as citizens (representatives of the general population) during the focus group sessions, but as consumers during CV surveys. The choice of the CV method presumes that the most important decision-making criterion is the economic welfare of the general population considering the given environmental improvement. In other words, the actors or stakeholders that could be the most affected by the decision taken (e.g., local farmers, river bank owners, *etc.*) do not get a voice as such in the process, which contrasts with the other cases considered.

In case B, the roles assumed by the participants were unclear, as is typically the case in deliberative monetary valuation. Participants were citizens, selected following stratification rules and they were asked to act as representatives of the general interest. They were given information reflecting socioeconomic data of the general population for the elicitation of WTP. The citizen framing implicitly refers to participants as social individuals, able to make value judgments, face hard choices and debate a political issue. However, the elicitation of WTP induces participants to respond as consumers. The authors note that some jurors had trouble understanding the notion of individual WTP and report that some conflict emerged between the citizen’s jury framing and the objective of eliciting individual WTP.

In case C, participants consisted of a variety of stakeholders, including federal, state and city decision-makers, public facility representatives, mining, energy and fish-farming industry representatives, a state-owned restoration company, and other civil society actors such as farmer and local tourism associations. Stakeholders therefore included both different categories of representatives of the political authorities in charge and other groups of economic actors and residents connected to the issue at hand. Participants had a clear idea of who was involved and why: The specific interests of stakeholders involved were framed and displayed through the impact matrix. Considering the valuation criteria, the economic language was dominant in the study. The authors of study C explain that this was the result of a consensus that emerged between the stakeholders and that the methodology allowed the positions of the affected parties to be made more explicit and to eventually take into account the interests of the least favored groups.

In case D, participants were also stakeholders, but the jury was only composed of five resource managers, chosen because of their involvement in another ecosystem services valuation project in the region, as well as in the development of a management strategy regarding tourism. Case D was unique in that the jury was composed entirely of formal decision-makers working within the institutional structures in charge of management. Some particularly affected stakeholders (e.g., the local ski resort representative, the water management authority, *etc.*) were heard by the jury as witnesses, but the process does not guarantee that all relevant stakeholders were heard and does not aim to represent citizens.

We can conclude that the type of actors involved and the role models assumed by the participants in the cases study vary greatly. In cases A and B, participants assumed both the role of consumers and citizens, which created confusion. Consumer and citizen premises imply to give a stronger voice in the process either to the economic benefits of the general population, or to the opinion of the general population, respectively. By contrast, stakeholder premises often entail stronger consideration of the interests of the affected parties. In both cases C and D, the participants were stakeholders, but the array of participation was nevertheless very different.

3.3.2. The Differences in Contribution between Participants

In case A, differences in contribution are justified by the logic of the valuation: The role of the citizens and experts involved in the focus groups was limited to the design of CV surveys and to the choice of the ecosystem services indicators. During the CV assessment stage, however, the differences in contribution between respondents were low because all participants were confronted with the same questionnaire. We therefore conclude for a medium level of differences in contribution in case A.

In case B, the jury constituted the heart of the assessment. For this reason, attempting to maintain a sufficient level of equity in contribution among participants during this stage was very important, and the authors paid a serious attention to this issue. Indeed, they argue that, in theory, deliberation should guarantee "*equal standing and effective voice*" for every citizen involved (p. 237). However, they also recognize that this ideal was not achieved during the study. Various sources of inequality in contribution are discussed (e.g., social and cognitive skills and capacities, prior knowledge, *etc.*), and this issue is viewed as fundamental for participatory processes in general.

In case C, the different groups of involved stakeholders contributed unequally throughout the entire process. Indeed, one particular group of stakeholders, the cross-state group, composed of representatives of local and national political authorities, had a dominant role. This stakeholder group was the only one involved in the workshops in which climate modeling was discussed and policy strategies and criteria were designed. It was also the only group of stakeholders to be involved in the single-criterion assessment. However, an important feature of the participatory study is that all stakeholders, including farmer and local tourism associations, as well as mayors of small cities near the river, *etc.* should normally take part in the decision-making process during the final MCA stage. It is highly unlikely that those actors would have been involved in decision-making without the implementation of study C. The issue of equity in contribution is seriously taken into account throughout the valuation process and is discussed in-depth by the authors in case C. During the final deliberative MCA assessment, the process should structurally guarantee equity in contribution; individual impact matrixes should be presented and discussed to attain a consensus over weighting. Therefore, we assign case C a medium level of equity in contribution considering the overall process: The large array of participation implied that this issue was more difficult to handle, although the MCA structure and the deliberative setting provided great help.

In case D, all of the stakeholders participated to the decision-making process. Equity in contribution was insured through the MCA structuring: Individual impact matrixes were presented and discussed with the purpose of reaching consensus over the appropriate weighting and policy strategy. Furthermore, during the second visualization of results, each juror was asked to justify their positioning and to explain the reasons underlying his/her weighting. We conclude for a high equity in contribution in case D, to which the deliberative process and the MCA structuring contributed. However, compared with case C, the higher degree of equity in contribution did not arise from methodological differences, but rather from the choices made regarding who was involved in the valuation and to existing differences between the institutional contexts in which the valuation processes took place.

Considering the process as a whole, both studies B and D managed to achieve a rather low level of differences in contribution, while we conclude for a medium level in studies A and C (for very different reasons). Deliberative MCA presents great advantages to deal with this issue, especially during the assessment phase.

3.3.3. The Level of Participatory Impact

CV typically implies consultation. In case A, the objective was to inform the political structures in charge of management on the efficiency of public investment in riparian restoration in the area considered. The inclusion of focus groups did not increase the level of participatory impact: Participants involved in the focus group were not consulted for problem scoping or for the design of policy alternatives, but only for CV survey design. A specificity of case A is that participants in both the CV surveys and the focus groups were treated as external subjects who were to be analyzed from a "neutral" point of view, and whose contribution was required only for assistance and data gathering. This contrasts with the other cases in which participants were treated as political individuals, able to make choices and to be further involved in the decision-making process.

In case B, the level of participation impact is hard to assess because of the fictional character of the study and because of the ambiguities of deliberative monetary valuation. Indeed, the authors state that the deliberative monetary valuation exercise was designed to provide results, which could be included in a CBA. For instance, it means that jurors followed a "*purchase model*" and not a "*contribution model*" (p. 238): They were not told how much the environmental improvement would cost when confronted with the WTP question. The objective of producing results to be used in a CBA implies that the goal of the study was rather providing information for the institutions in charge of management than making a political choice about the management options considered. This explains why we consider that the level of participation impact in B was consultation, as in case A. However, through the deliberative framework, participants had a higher degree of control over the outcomes of the process compared with A. Furthermore, participants were treated as political individuals able to face hard choices, information flows were very interactive and participants had to reach a consensus over a preferred management option during the first part of the exercise. The status of the deliberation (*i.e.*, the qualitative insights about the preferences of participants regarding the various management options considered) remains ambiguous in case B: Should the debates preceding WTP elicitation be considered instrumental (*i.e.*, viewed as a "trick" to elicit socially constructed WTP), or would their contents have mattered *per se* for decision-making?

In cases C and D, the level of participation impact was collaboration because both processes were very interactive and information flows were shared between participants. Furthermore, participants had a strong control over the outcomes of the process and a strong influence on both policy alternatives and on the decision *per se*.

We can conclude that the level of participation impact was clearly stronger in cases C and D than in case A and B, although the latter remains ambiguous regarding this point.

3.3.4. The Level of Democratization of the Decision-Making Process

In case A, the history of the environmental problem considered is well documented, and the study concludes for both the scientific community (the dimension of scale can be taken into account in CV and CBA as an indicator of ecosystem services provision) and for the institutional decision-makers in the region (the optimal solution is full restoration, which has the best cost-benefit ratio). The focus groups were only implemented as an aid to the expertise of the analysts and to assist the research team. Institutional structures in charge of management are understood as sovereign decision-makers who can decide whether to take into account the outcomes of the study. The goal was not to provide opportunities for the local population to be further engaged in environmental decision-making, and the level of democratization of the decision-making process is therefore low.

In case B, the level of democratization of the decision-making process is unclear. Indeed, the authors state that one of their objectives is "*incorporating community attitudes and values into decision-making*" (p. 225). Topics such as participatory democratic theory and deliberative and discursive democracy are discussed. However, at the same time, the process was framed to provide a sounder elicitation of WTP estimates at the service of external political bodies, in a configuration close to that of case A. The citizens' framing presented an opportunity to democratize the decision-making process and to foster the level of participation impact. This would have implied to discharge, at least partly, the management authorities of their decision-making power. For instance, their implication in the study as witnesses would have enforced the legitimacy of the process, but the decision-making would have operated beyond their reach. However, in that perspective, what could motivate the political authorities to participate in the process, and does monetary valuation still have a purpose? This indicates a problematic aspect of deliberative monetary valuation: Tensions arise between the normative claims of deliberative democracy and the goal of providing useful economic estimates for existing institutional structures. It also raises questions regarding the engagement of political authorities in deliberative valuation (e.g., are the political authorities willing to engage, why, how *etc.*).

In case C, the authors insist on the need to offer "*practicable science-based decision support processes*" (pp. 63–64) for environmental problem solving. The goal of the study was to improve decision-making in terms of competence and fairness through the development of a participatory process. The implementation of participation initially relied on the will of the political authorities themselves, who implemented a preliminary participatory initiative, from which the formation of the cross-state group resulted. Because of study C, however, the diversity of the stakeholders involved in the decision-making process increased greatly. On the one hand, the fact that a specific group of stakeholders, the cross-state group, played a dominant role in the process contrasts with the ideal of deliberative democracy and with the objective of transferring decision-making power to citizens. But on the other hand, study C fostered the will and the capacity of the political authorities to engage in a large participatory process. We therefore conclude for a medium capacity of democratization.

By contrast with case C, in case D all of the participants were representatives of the political authorities in charge of management. The main outcome of the study for decision-makers was not only the confirmation of a policy strategy but also the insurance that the degree of consensus over the choice of this strategy was higher at the end of the process than before. However, we conclude for a low degree of democratization of the decision-making process: The study was not designed to allow the inclusion of all affected or interested stakeholders, or to foster the engagement of citizens. Some affected stakeholders intervened as witnesses, but they did not took part in decision-making. We can conclude that from all the cases considered none attained a high level of democratization of the decision-making process. This dimension varies across studies for different reasons. Cases A and D did not aim at fostering this dimension because they were designed as decision-aid tools, which contrasts with cases B and C. From the latter we can conclude that designing processes involving political authorities, while trying to foster the engagement of other stakeholder groups in decision-making is a delicate operation. However, case C illustrates that deliberative MCA can provide a great help in that task. Regarding case B, we can conclude that deliberative monetary valuation proved ambiguous: The purpose of involving citizens in decision-making contrasts with the objective of producing a monetary estimate.

4. Conclusions

Aligned with previous research results [9,14,15,17], our analysis confirms that non-deliberative and deliberative participatory methods are relevant means to address the complexity involved in environmental valuation and to reduce the shortcomings of traditional decision-making methods. In the cases considered, participation often helped participants to address the cognitive, normative and composition problems. Overall, deliberative multicriteria evaluation showed a great potential to address cognitive and normative issues because it allows the problem structure to be made more explicit and salient in the minds of participants, and provides time for preference construction without necessarily forcing trade-offs across value dimensions. However, even combined with deliberation, the cognitive and normative problems remain for monetary valuation (and especially for stated preference methods, such as contingent valuation, which was analyzed through the cases considered). Regarding the composition problem, we note that the design of indicators often tends to remain strongly dependent on the choices made by the research team and on the involvement of experts. However, cases C and D showed that highly interactive processes involving deliberation have good potential to address this issue, especially because they produce a collective reflection on the matter and allow for the chosen indicators to be adapted to the needs and wills of the participants involved, while fostering their understanding of the various dimensions of the problem under consideration.

Participation may therefore help to address the complexity involved in environmental valuation, and yet it nevertheless covers a great variety of processes, associated with specific purposes, which can influence all dimensions of value articulating institutions (who is involved, what counts as data, how are they articulated, *etc.*). The quality and legitimacy of the outcomes of valuation processes become strongly dependent on how participation is framed. Based on the literature and the analyzed case studies, we showed that the roles assumed by participants, the differences in contribution between participants, the level of participation impact and the level of democratization of the decision-making process are important criteria useful to assess complex valuation processes, combining non-deliberative and deliberative participatory features with classical methods such as CV, CBA and MCA. Our analysis suggests that those criteria are strongly interdependent, but not systematically positively correlated. Furthermore, we note that those issues are strongly related to the institutional and political contexts in which valuation studies take place.

Indeed, our analysis suggests that both cases A and D were designed as decision-aid tools, aiming at providing expertise and scientific structuring for a decision-making power that remains sovereign. By contrast, in case C, some stakeholders who were not institutional decision-makers had a stronger influence on decision-making during the study. Case B remains ambiguous regarding this point: The involvement of citizens offered a great opportunity to increase the level of participation impact and the level of democratization of the decision-making process, but the valuation was framed to produce a monetary estimate useful for the institutional authorities in charge of management, whose decision-making capacity has not been further engaged in the valuation process. What contrasts between cases A and D, however, is that in D participants and decision-makers (*i.e.*, both the jurors and the witnesses) were treated as political individuals upholding values as well as particular interests and capable of reasoning and arguing.

Participatory and especially deliberative environmental valuation methods therefore still face numerous issues: Designing processes involving both citizens and stakeholders, including representatives of political authorities in charge, in a situation of conflict is a delicate operation. From a theoretical perspective, the field of participatory environmental valuation seems to be at a crossroads between, on the one hand, the normative claims of deliberative democracy advocating the empowerment of citizens as a means for values transformation towards sustainability, as well as the necessity to valorize political forms of decision-making, *i.e.*, the expression of value judgments and arguments, and, on the other hand, the need to develop technical tools and processes aiming at fostering the engagement of political authorities and institutional structures in environmental decision-making. Indeed, the engagement of political authorities can be seen as a necessity, in order to insure that the results of valuation studies have an impact in "real-life", but it can also constrain the ability to effectively democratize decision-making.

Our analysis also shows that if participation is generally regarded as a means to reduce technical shortcomings of methods, such as CV and MCA, implementing participatory, and especially deliberative processes, implies considering core institutional and political issues, such as who has the ability to make a decision during the process, at which conditions the political authorities in charge are willing to engage in participatory valuation processes, how to insure that the outcomes of the valuation process will effectively be taken into account within decision-making, *etc.* Therefore, the effectiveness of environmental valuation tends to become increasingly dependent on contextual and political dimensions.

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Author Contributions

Each co-author of this paper contributed equally to the design and the data analysis of the reported research. The first author led the development of the final version of the analytical framework and the writing of the paper.

Conflicts of Interest

The authors declare no conflict of interest

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Article

Market-Based Instruments for Ecosystem Services between Discourse and Reality: An Economic and Narrative Analysis

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Abstract: Since the mid-1990s, the concept of ecosystem services has become increasingly popular in academic circles and among decision-makers. Because of its inclusive character, this concept has given rise to different interpretations in economics. Since its inception, it has been associated with the development of market-based instruments (MBIs) in conservation policies. From this perspective, the sustainable provision of ecosystem services is hindered by market failures (e.g., public good attributes, externalities) and prices that do not capture the full value of the natural assets. MBIs are therefore recommended. According to their promoters, they provide powerful incentives to conserve the environment while at the same time offering new sources of income to support rural livelihoods. Our paper contends that different economic narratives, and associated representations of the market failure at stake with the provision of ecosystem services, may support different policy instruments that are all coined as MBIs. As an illustration, we analyze the economic discourse underlying payments for ecosystem services and eco-labels, and we underline the variety of institutional forms to which they give rise in order to emphasize the differences between discourse and practice.

Keywords: market-based instruments; ecosystem services; externalities; joint products; narratives; payments for ecosystem services; environmental certification

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1. Introduction

Since the mid-1990s, the concept of ecosystem services (ES) has become increasingly popular in academic circles as well as among decision-makers [1]. The number of articles on ES in the international databases Web of Science and Scopus has multiplied by seven in 10 years [2]. Beyond the academic literature, a range of books and reports arising from international initiatives have been distributed and disseminated, such as the United Nations' Millennium Ecosystem Assessment [1], *The Economics of Ecosystems and Biodiversity*, a report initiated by the G8 [3], and the *State of Food and Agriculture 2007* [4]. Several networks have also been launched since 2000 to promote and mainstream this concept into conservation policies and agendas. Some focus on the economics-environment nexus (Ecosystem Valuation, Earth Economics, Earthtrends, *etc.*), others are devoted to awareness-raising (Guardian Environment Network, Business Green, Ecoworldly, *etc.*), and still others (Conservation Finance Alliance, Katoomba group, Ecosystem Marketplace, Avoided Deforestation Partners, BBOP Learning Network, Nature Valuation, and Financing Network, *etc.*) focus on conservation funding. ES are the benefits people obtain from ecosystems [1]. The services provided by ecosystems sustain or protect human production or consumption activities or affect welfare in general. They are classed into four categories: provisioning services, regulating services, supporting services, and cultural services. They include material and non-material benefits derived from ecosystems in their natural state or modified by human practices.

Economics has played a major role in the emergence of the ES concept as an explicitly anthropocentric concept and has contributed to its dissemination and politicization [5]. However, beyond conjuring up images of a nature bent toward human welfare, this concept is still open to multiple interpretations in ecology as well as in social sciences [6–8]. The risk of this cross-disciplinary notion being turned into a hackneyed phrase, supporting neoliberal discourse and the re-labeling of public policy provisions as market-based instruments (MBIs), is often highlighted [9]. Indeed, the concept of ecosystem services has gained momentum and has gradually replaced mentions of nature and biodiversity in the formulation of environmental policies at the very time when the latter were overtaken by a neoliberal discursive contagion epitomized in the widespread adoption of the term "MBIs".

The definition of MBIs is very broad and comprehensive and this notion has prompted criticism along the same lines as ES. While referring to the market, this notion is not clearly anchored in economic theory. Its definition and scope are still debated, and the term is very loosely used to refer to a wide range of policy tools such as taxes, cap-and-trade allowances, eco-labels, carbon or biodiversity offsets, or payments for ecosystem services [10–12]. There is no consensual list nor commonly accepted classification of MBIs. They are defined through their opposition to command-and-control instruments. The market mechanism is used to set incentives via prices or quantities (or contracts). MBIs can be classified into three different groups: price-based, quantity-based, and market friction mechanisms [13]. The context in which they might be effective has been discussed in the literature [14] and their social, political, and ideological underpinnings have given rise to criticism [15]. Our chief interest here does not lie so much in these issues raised by MBIs in general as in MBIs in relation to ES, since these notions have united and mingled in the recent policy discourses they have come to epitomize.

The purpose of this paper is to explore the major economic narratives that are relied upon to legitimize MBIs for ES and to compare the expected features and outcomes of these institutional arrangements to their existing forms. After an account of the emergence of the ES concept, we review different interpretations and meanings attached to this concept in economics (Section 2). We show that economists legitimize market-based policies to protect ES by conceptualizing ES either as positive externalities or as joint products or services, for instance, of agricultural commodities. Then, we focus on mechanisms that are the most obviously associated with given economic representations of ES: payments for ecosystem services (Section 3) and eco-certification schemes (Section 4). This is followed by a discussion of the innovative nature of payments for ecosystem services (PES) and eco-labels as instruments for the provision of ecosystem services. We show the gap between the legitimizing narratives supporting these policy instruments and the real-life institutional arrangements in which they are embodied.

2. The Concept of Ecosystem Services: From Inception to Economic Reception

The publication of the *Millennium Ecosystem Assessment* synthesis report in 2005 was a key stage in the career of the ES concept. Its exposure since the turn of the century can best be analyzed in light of the origin of the concept in ecology and economics.

It has long been acknowledged that ecosystems play a positive role in human well-being. Daily [16] notes, for instance, that Plato's *Critias* refers to the role of forests in erosion prevention and watershed development. However, the term "ecosystem services" itself did not appear before the beginning of the 1970s, in relation to the development of environmentalism in the United States [8,17] and to the neo-Malthusian theses of Paul and Anne Ehrlich [18,19]. These authors tried to raise awareness about the alarming pace of resource exhaustion, species extinction, and ecosystem conversion. To that purpose, they emphasized human dependence on nature and developed global ecology and ecosystem approaches [20]. They strove to demonstrate the critical contribution of the various services provided by ecosystems to human welfare.

The limitations of standard environmental economics to account for this dependence on nature were stressed from the 1980s by a group of economists and ecologists and led to the creation of ecological economics. In a plea for this new approach, Costanza and Daly [21] stated as their proposed objective to "envelop economics in an overall ecological framework" (p. 7). This need for a theoretical and methodological renewal was felt by several authors who concentrated on the development of an adequate analytical framework during the 1990s (e.g., [22,23]). The concept of ES was instrumental in this regard. Picturing biodiversity erosion as a threat for the provision of critical services and as a net loss in terms of welfare made the issue more tangible.

A further step toward the affirmation of the links between ecosystem conservation and economics was taken at the end of the 1990s with the publication of Daily's book [16] and of the article of Costanza *et al.* [24] in *Nature*. For allegedly pragmatic reasons, these authors defend the monetary valuation of ecosystem services, arguing that, to quote Myers and Reichert [25] (p. xix), "we don't protect what we don't value". Because monetary valuation is supposed to be particularly appealing for decision-makers, it has become a key element of the conservationist rhetoric and advocacy. Although it was a much-debated topic within ecological economics during the 1990s, economic valuation of the environment has now become commonplace. Economics has become the language for environmental policies precisely when ES became a hegemonic concept in the field of conservation policies. The ES concept has been both a driver and an outcome of this evolution. Biodiversity loss has been rephrased in terms of market failure, and market-based instruments for the provision of ES, coupled with the suppression of perverse subsidies, are presented as solutions to halt species loss and ecosystem degradation [26,27]. The former policies, being based on regulation and preservation models that limited or prohibited the exploitation of endangered species and their habitats, as well as participatory approaches, have been "marginalized".

The mainstreaming of the ES concept in policy-making at different scales (international, national, local) is a cornerstone of the "new economy of nature", to paraphrase Daily and Ellison [28]. This is the main objective of the approach followed by The Economics of Ecosystems and Biodiversity (TEEB) initiative (2007–2012), which "calls for a change to the current economic paradigm; at the same time (…) acknowledges the persuasive power of economic reasoning in contemporary societies" (Ring *et al.* [29] (p. 15)).

The economic characterization of ES used to justify conservation builds on two major representations: ES as positive externalities (hereafter the "externalities narrative") and ES as joint products (hereafter the "joint products narrative"). The former narrative has given rise to dedicated market-like arrangements as ES became a matter of political concern. The latter narrative has led to the reconsideration of some ES (such as most provisioning services) that were already, at least to some extent, commoditized or associated with existing markets. Both of these are discussed below.

At the core of the "externalities narrative" is the two-fold definition of externalities (Baumol and Oates [30] (p. 17)):

> (1) "An externality is present whenever some individual's (say A's) utility or production relationships include real (non-monetary) variables, whose values are chosen by others (persons, corporations, governments) without particular attention to the effects on A's welfare. (...) It should be noted also that this definition excludes cases for which an individual deliberately acts to affect the welfare of A."
> (2) "The decision-maker whose activity affects others' utility levels or enters in their production functions does not receive (pay) in compensation for this activity an amount equal in value to the resulting (marginal) benefits or costs to others."

Considering ES as positive externalities implies that the proposed policy program to ensure their socially optimal provision is to have them internalized. In line with economic theory, this can be achieved through a transaction between the beneficiaries and the providers of the services in question, which may take different forms and either be direct or imply third parties. Various options can be followed: taxes or subsidies, contractual public policies, or direct contracts between service providers and beneficiaries (actually the whole range of environmental policy tools, from Pigovian taxes to Coasian contracts, through quota markets). The "externalities narrative" thus legitimizes and gives rise to a range of various mechanisms that have been grouped under the common heading of market-based instruments since the end of the 1980s [31]. We will focus on one of the most common market-based instruments in the context of ES provision, namely "payments for ecosystem services" (PES) schemes, in the next section to emphasize the controversies and the differences between narrative and practice.

The "package economy" approach [32] is the basis for the "joint products narrative" regarding ES that leads to the selection of other market-based instruments, such as environmental certification, as possible tools for ecosystem management. In this perspective, ES are considered bundles of functions (or attributes) that might be included in or attached to physical goods, following Lancaster [33]. Contrary to externalities, these bundles of functions are produced purposely, they are part of a differentiation strategy and are expected to generate rents. Beyond their intrinsic material features, products such as agricultural commodities are considered as repositories of values. They can be analyzed as "packages". For instance, they might encompass special commitments relating to the organization of the supply chain (about animal welfare, the use of agrochemicals, ethics, *etc.*). These are not, strictly speaking, "joint products", as described in economic theory, but rather "joint services". Recent years have witnessed an increase in the informational or immaterial content of agricultural goods to address rising concerns about health or the environment. These goods are sometimes referred to as "agricultural solutions" to stress that this change is driven by consumer demand and implies immaterial dimensions. What is produced and sold is a package of complementary services and goods [32]. For instance, certified shade-grown or bird-friendly coffee is cultivated and purchased with a view to protect the environment and to provide ES. It commands a premium price because it includes "services". Certification by an independent third party guarantees respect for specifications relating to the provision of specified ES. The higher prices commended by certified products on the markets could be considered evidence of the consumer's consent to pay for ES. From this viewpoint, eco-certification schemes—insofar as they relate to identifiable ES—can be considered market-based instruments for the provision of ES. This line of justification for eco-certification schemes has, however, been limited thus far. As we will show, the link between eco-certification and ES provision is tenuous and more difficult to substantiate than in the case of PES mechanisms.

The controversies and theoretical debates raised by these narratives can best be illustrated by critically analyzing the institutional arrangements to which they give rise, PES that are associated with the "externality narrative" (Section 3), and eco-certification issuing from the "joint products narrative" (Section 4).

3. Payments for Ecosystem Services: From Discourse to Political Realities

The "externality narrative", which is conveniently familiar for standard environmental economics, has been favored so far in the justification of MBIs in relation to the provision of ecosystem services. In line with this narrative, ES are internalized through instruments that are given the generic name of payments for environmental services or payments for ecosystem services (PES). The implementation of these mechanisms is therefore often justified by the existence of positive externalities (to be encouraged) or negative externalities (to be cut) induced by production activities (agriculture or forestry). They can allegedly provide powerful incentives to conserve the environment, while at the same time offering new sources of income to support rural livelihoods [34]. In the following, we will first analyze argumentative patterns and blind spots in the economic PES discourse, and then confront the PES discourse with its political realities in developing and industrialized countries. Indeed, PES schemes can take different forms according to the context, they seldom are true markets in the real world, and the promotion of PES often reflects a requalification of pre-existing public intervention systems.

In PES schemes, the beneficiaries or buyers of environmental services compensate or remunerate those who provide (or rather contribute to the provision of) the service. According to the "canonical" definition proposed by Wunder, PES are voluntary transactions conditional upon clearly defined environmental services between a provider and a beneficiary [35]. They must be conditional (*i.e.*, user payments are contingent upon the service being effectively provided) and additional (*i.e.*, they generate a higher level of ecosystem services than in a baseline scenario, without payment). According to their promoters, they reconcile individual and/or collective land use decisions with social goals in terms of natural resources management and biodiversity conservation. Actually, most policy instruments that have at least some of the distinctive features of what is held to be the "archetypical" PES are re-labeled as such. Therefore, real-life PES seldom meet all of these requirements, as aptly noted by commentators [36–39]. PES have been described as Coasian-type contracts to ensure the provision of ES. However, unlike pure Coasian negotiations, the parties involved in the transaction are not free to determine who pays whom. The PES discourse is not neutral in this regard. Most of the time, it assumes a beneficiary-pays (rather than polluter-pays) principle and, therefore, an implicit distribution of rights and correlated duties over resources and ecosystem services [40]. Indeed, the rights of so-called "providers" of services, whose activities favor conservation or who refrain from destructive practices, are given precedence over those of the beneficiaries, who must pay to have their claims to ES acknowledged. The frequent references to the Coase theorem in relation to PES tend to obscure the issues raised by this allocation of rights. Indeed, according to the Coase theorem, the initial distribution of property rights is neutral in the final outcome of the negotiation only on the condition that rights can be reallocated at no cost. This is definitely not the case with PES because laws or regulations define the entitlements of the parties involved in these mechanisms.

PES schemes do not fit within the Coasian ideal, and their economic characterization is therefore trickier than what might appear at first glance. The problems of defining the exact nature of the transactions involved in these mechanisms are reflected in the words used to describe them. Indeed, there is no widely accepted designation, and at least four different terms (payments, markets, rewards, and compensation) are used in this context [41]. The most frequent and generic word used in relation to PES is "payment", which is more a layman's term than an economic concept that could be associated unambiguously with a theoretical model. It implies a monetary transaction but does not rely on specific assumptions about the liability and property of the parties involved, nor does it apply to externalities. Although the most neutral word to describe transactions in relation to ES, it can still create ideological conflicts and some obstacles to the implementation of well-intentioned PES schemes. For instance, the project of the Fundación Natura Bolivia financed by Forest Trends, which is referred to as a PES mechanism abroad [42], is defined as a "reciprocal watershed management" project in Bolivia to prevent the use of the term "payment" in a spirit of conciliation in line with the country's political position against the commodification of nature [43]. The term "market", which has recently emerged in connection to ES (Markets for Ecosystem Services/MES), is obviously used to suggest their efficiency in referring explicitly to economic theorizing. It falsely conveys the idea of competition among a multitude of buyers and sellers, whereas in reality there are often direct contractual relations among very few beneficiaries and providers, without centralized information. Moreover, it can prove a double-edged sword: in developing countries, the term "market" is often associated with a threat of privatization and commodification of services that were freely or cheaply available [41]. The other terms used to describe PES express a shift towards stakeholders and their practices as justification for the transaction. "Reward" is reminiscent of merit, justice, and fair remuneration. It implies that the recipients have taken positive actions to supply ES, which is at odds with the presentation of the latter as externalities, e.g., unintentional outcomes of economic activities. Moreover, because service providers are rewarded even if providing services does not cost them anything, it may lead to conflicts when the environmental outcomes of the PES do not live up to the expectations [40]. Finally, the term "compensation" is also used. In such a perspective, the direct and opportunity costs supported by the service providers to fulfill their commitments under the PES system, e.g., change in location or practice, substantiate their claims for compensation [40]. In that case, the rationale for the transaction is not ES as such but the costs induced by the environmental policy, which is therefore treated as an infringement of prior rights and could itself be considered an externality. These terminology issues demonstrate that the internalization process at stake in PES, if any, is complex and subtle.

Beyond semantics, real-life PES mechanisms take various forms depending on countries. First, there has been a multiplication of PES contracts since the turn of the millennium in developing countries [44]. These contracts cover a very broad spectrum, ranging from national programs managed by governments to local projects of more limited scope funded by the private sector, NGOs, or cooperation agencies. The different elements of Wunder's definition of PES apply more or less depending on the number and type of ecosystem services involved, the payment mechanisms used, and the number of buyers and providers involved in the transaction [27,35,45]. The Costa Rican Pago Por Servicios Ambientales, which was established in 1996, is showcased as a flagship PES-labeled scheme. It provides payments (more precisely "rewards") to landowners according to their land uses—forest conservation, reforestation, sustainable management, *etc.*—with the justification that these land uses generate ES either locally or globally [46]. This program does not meet Coasian criteria, and it reflects a requalification of pre-existing public intervention systems, particularly in forest policy. Indeed, it encompasses and redefines the former system of subsidies implemented by the Costa Rican government to fight deforestation. One explanation for this situation is the signing in July 1995 of an agreement between Costa Rica and the International Monetary Fund (IMF) banning the Costa Rican government from paying subsidies to productive sectors. In that context, a new type of justification had to be found for supporting the forestry sector, shifting from a government support rationale to market-based instrument rhetoric [47]. This agreement was part of the negotiations for Costa Rica's entry to the WTO and negotiations between Costa Rica and the World Bank for a structural adjustment plan [48].

In industrialized countries, the most advertised PES are those based on voluntary contracts, which arise from "self-organized" bilateral negotiations after the Coasian model. One of the most often-cited examples is the Vittel case in France: the company has signed contracts with surrounding farmers whereby they commit themselves to either change their practices or give up their production in exchange for payments in order to maintain the quality of mineral water [49]. Aside from this emblematic case, the promotion of PES is usually reflected in a requalification of pre-existing public intervention systems, particularly in agricultural policy. Their purposes are relabeled in terms of promoting ES in a market-based or market-compatible fashion. This is particularly the case with certain agri-environmental measures established in Europe and the United States [50,51]. References to the provision of ES tend to replace earlier justifications based on rural development and the multi-functionality of agriculture [52,53], which featured prominently in European debates on reforming the Common Agricultural Policy. ES are a new discursive resource used to legitimate agricultural subsidies and support measures for farmers that would be challenged otherwise because of the distortions they are likely to generate for competition. Re-qualifying them as PES makes it possible to present them expressly as market-based instruments and not as protectionist tools while at the same time tapping into an international discourse (ES) and hence avoiding the European Union language register (e.g., the term "multifunctionality" and its reminiscence of the Common Agricultural Policy). Nevertheless, this tentative ecologizing of agricultural subsidies is not completely successful; the ES/PES debate is still viewed with distrust by developing countries (agricultural exporters) that perceive it as just another attempt to defend protectionist interests. Worldwide, Coasian contracts seem to be an exception rather than the rule in the organization and functioning of PES. The description of ES as externalities is most likely, in some cases, a specious argument to reinforce policies that are considered irrelevant and inefficient according to neoliberal standards and that are threatened as such.

Finally, the relevance and the results of PES are also controversial. According to Wunder [35] and Laurans *et al.* [34], PES as MBIs have great potential for halting the degradation of natural resources, attenuating the imperfections and limited successes of integrated conservation and development projects (ICDPs) or sustainable resource management, and mobilizing additional financial resources. They would also create opportunities, especially in developing countries, including diversification of incomes and activities, job creation, and capacity-building [14,54–56]. These are the theoretical advantages attached to market-like solutions according to the externality narrative. However, the analysis of real-life PES mechanisms leaves some doubt about their effectiveness, *i.e.*, their ability to meet contractual environmental objectives. Their efficiency, their fairness, their legitimacy, and their sustainability are also questionable, as illustrated by Muradian *et al.* [11,36] and Legrand *et al.* [57]. Indeed, the contexts in which PES are developed are often characterized by imperfect and asymmetric information (scientific uncertainty, inadequate ecological knowledge, inappropriate methodology for controlling the status of environmental services, *etc.*) and imbalance of power, allowing strategic behaviors such as hijacking and appropriation of the instrument by stakeholders who were not initially targeted and funding capture. According to Coase himself, these are the very features that should exclude the development of contractual agreements along Coasian lines as internalizing instruments. Not surprisingly, such contracts sometimes result in the weakening of public authorities and policies, a degradation of ecological systems, limited innovation in sustainable practices, the commodification of biodiversity, and a worsening of inequalities. There is definitely a gap between the legitimizing economic discourse supporting PES development and the political and environmental realities in which it is applied.

From their very inception, PES mechanisms have been meant to provide for the supply of ES. It is therefore quite normal that they should appear as the privileged type of MBI to account for ES. However, environmental certification is sometimes referred to as well in this context [10–12,58,59].

The economic discursive devices underlying the re-designation of eco-certification as MBIs and their factual accuracy both deserve investigation.

4. Eco-Certification: Recognition of Ecosystem Services as Joint Products?

Several types of environmental certifications are now re-labeled as MBIs for the provision of ES, although their initial purposes and justifications were different. New objectives and a new corpus of justification are now attached to these instruments that are thus redefined in the process. As already mentioned, the “joint products narrative” supports the presentation of eco-certifications as possible MBIs for the provision of ES. In the following, we will examine to what extent this narrative measures up when tested against real-life arrangements.

Certification is defined by Bass *et al.* [60] (p. 2) as “a procedure by which a third party provides written assurance that a product, process or service conforms to specified standards, on the basis of an audit conducted to agreed procedures”. Certification is justified by the fact it provides consumers with information and guarantees the characteristics of products [61]. It is particularly relevant when these features can neither be observed nor verified, either during the transaction or afterwards. These characteristics are called credence attributes. They may relate to the product itself or to the production process and processing techniques (impacts on the environment, labor conditions along the supply chain, *etc.*).

Eco-labels can be considered market instruments on several accounts. The producers voluntarily engage in certification and commit themselves to respect the associated standards. Eco-certification explicitly refers to the compliance with environmental criteria and standards. Certified products command a premium price that could be interpreted as a payment for the maintenance or supply of ecosystem services.

However, certification has not always been so clearly considered a market instrument. There have been progressive changes in this regard, especially over the last two decades. The early experiences of organic agriculture, *i.e.*, safer and more sustainable agrofood systems embedded in biological processes [62], relied upon public labels and took place in Europe and the United States. The creation of the International Federation of Organic Agriculture Movements (IFOAM) in 1972 has favored the mainstreaming of these experiences and the spread of organic farming to the south. The distinctive signs applied to organic products have also diversified. In most countries, organic farming is currently organized and governed through a combination of public regulation and private certification schemes [63]. Many trademarks and certification tools that specifically refer to what could be termed biodiversity or ecosystem services have developed from the end of the 1990s for various products. In the coffee sector, in addition to generic organic labels and eco-labels, there are a growing number of specific certification tools, e.g., Bird Friendly, created in 1998 by the Smithsonian Migratory Bird Centre; Utz Kapeh (changed to Utz Certified), created in 1999 by a group of European large retailers; Rainforest Alliance, created in 2003 by the Sustainable Agriculture Network (SAN); C.A.F.É. practices (Coffee and Farmer Equity practices), created in 2004 by Starbucks; the Common Code for Coffee Community-4C, created in 2005 by private companies and international organizations of the coffee sector and supported by German cooperations; and, more recently, Nespresso AAA certification, developed by Nestlé in 2006 [64–66].

The discourse legitimizing their use within the context of the ES policy mix has been built up as well during the last decade. The alleged advantage of eco-certifications as candidate incentives to support the provision of ES is that these tools already exist and are well known and recognized by consumers. They rely on existing markets, whereas the future potential of *ad hoc* exchange mechanisms specially created to account for newly identified and defined ES is mere conjecture. Moreover, environmental certifications have met with growing success, particularly over the last decade [67]. These characteristics are considered material advantages over other possible policy tools that could argue for their widespread adoption for pragmatic reasons. The joint products narrative presented above has therefore been developed as a justification for the integration of certifications in surveys of MBIs for ES. However, in contrast with PES, this inclusion has not been promoted by practitioners or specialists of global value chains, standards, or labels. It stemmed instead from scholarly endeavor and is associated with attempts to inventory, classify, or compare MBIs [10,12,68]. *Ad hoc* legitimizing narratives have been drawn up in retrospect. Not surprisingly, the interpretation of certifications as MBIs for ES provision can appear as contrived. The expectations attached to these instruments attest to a lack of awareness of their actual functions. The discrepancy between the narrative legitimizing such a categorization and existing institutional arrangements is much larger than in the case of PES, and while the influence of the former is undeniable, it is not readily acknowledged and voiced by stakeholders. There is no broad consensus on these new discursive registers and claims.

Real-life certification tools have some distinctive features that should lead to a reassessment of their representations as MBIs for ES [69]. It must first be stressed that it does not make much sense to speak about eco-certification in general. In some countries, standards are formulated and overseen by the government, while in others they depend on private sector actors, with each one developing its own set of criteria. The specifications, the environmental requirements they include, and their monitoring conditions may vary greatly from one certification body to the next and from one product to the next. Most of them do not imply major changes in the farming practices or the processing techniques. It is therefore unlikely that they should provide additional benefits for the environment if they were to become widespread. For instance, it is often noted that many agricultural systems in the developing world are *de facto* organic and that, in such cases, formal certification may bring limited technical changes.

Beyond the diversity of eco-certification schemes, they have some common features that make their presentation as market-based or market-like and their relation with ES debatable.

Notwithstanding the expectations associated with the "joint products narrative", it proves difficult to connect eco-certifications in an unambiguous way with specific ES. Eco-certifications are not defined in reference to the places in which the products originate. Most of the time, the specifications relate to a given product (e.g., coffee, cotton) and can apply anywhere, regardless of the local context and area of production, without adaptation of the standards. The certification criteria pertain to farming practices or processing techniques and not to their environmental impacts, which might, however, be place-specific and, in any case, are often poorly understood. While the link between land use and carbon storage is well established, the relations between land use and biodiversity conservation have not been fully investigated. Moreover, the link between land-use and water services is often difficult to demonstrate because of the complexity of the water systems [70,71]. Eco-certification is an instrument for differentiating the products on the market. The premium prices derived from certification depend on the consumers' willingness to pay, which depends in turn on their perception of the product attributes and specificities. Certification criteria must be transparent and easy to monitor, and they should appeal to laypersons (such as the bird-friendly certification) rather than be based on accurate but complex and subtle ecological knowledge that cannot be summed up in a few simple indicators. The premium prices paid in this context can be considered, at best, very rough proxies of the value attached to the protection of the environment.

Presenting the premium prices paid for certified products as the willingness to pay for ecosystem services is therefore questionable. The higher price the producers receive for their certified products is, foremost, a means for them to cover the costs induced by certification and the prior formalization and standardization of processes and operations that were informal before. Furthermore, in practice, the producers, producing countries, and various types of labels and certifications are competing with one another, which might induce a downward trend in the premiums in the long run [72].

Finally, the benefits of eco-certification for the farmers and, hence, the assumed compensation they would receive for the ecosystem services they provide and the incentive it would create, are debatable. The only producers who can benefit from a premium price are those who comply with the environmental specifications, but there is little evidence of the reverse. Despite a growing demand, the supply of most certified products is still greater than the market outlets [73]. Even if they comply with the specifications, the producers are not guaranteed to sell all of their products at higher prices in certified marketing chains. Due to low demand, they might be obliged to sell the bulk of their products on the conventional market—at a price that does not account for the specific conditions of production. The distribution of the premium associated with product differentiation might not benefit the farmers. Downstream actors of the value chain might capture the differentiation rent, as has been demonstrated for the coffee value chain, which is dominated by roasters and retailers [74]. The farmers who have changed or adapted their practices to supply environmental services are not fully compensated for their efforts, while other actors who did nothing can use their position of strength to capture premiums. The rent distribution along the value chain is governed by the balance of power.

The interpretation of certifications as MBIs for ES tends to obscure the real nature and complexity of value chains and to create the appearance of a direct contractual negotiation between ES providers and beneficiaries, whereas many actors with various statuses are involved.

5. Conclusions

As we have shown, since its inception, the notion of ES has been associated with the development of market instruments in conservation policies. From this perspective, the sustainable provision of ES is hindered by market failures (e.g., public good attributes, externalities) and prices that do not capture the full value of the natural assets. Depending on how these issues are defined and prioritized, different types of instruments, implemented through different types of institutional arrangements, are suggested as policy tools. The craze for market development has also led to rethinking and rewording existing policy instruments as MBIs. It has even encouraged a shift in these arrangements, instilling market attributes into them.

A wide range of policy instruments, such as PES and eco-certification, reflecting various purposes and involving different actors are therefore presented as MBIs. They are characterized as such not only because of their inherent properties but also for the promise they show. The economic narratives that justify their adoption and development build on alleged rather than actual characteristics, associated with theoretical archetypes rather than existing institutional arrangements. We have emphasized the differences between discourse and practice. Considering policy tools as market-like or market-based tends to obscure the power relations underlying them and the regulatory framework within which they often take place. The contracts are presented as voluntary and are supposedly defined through mutually agreed terms, but in practice the law may restrict them. Similarly, the part played especially by state actors in the enforcement of so-called MBIs is often overlooked; they are presented as third parties, intermediaries, or brokers to fit within the Coasian ideal of bilateral contracts, but their influence is crucial. Finally, relabeling policy instruments as MBIs for the provision of ES does not negate their earlier organization and goals, e.g., the support of rural income and of small-scale family farming in marginal areas. The latter may impede the use of these tools for the promotion of ES. Indeed, while not completely unconnected, the pursuit of redistributive justice and local development and concerns for local environment protection may require distinct approaches and priorities.

The economic discourse on MBIs for ES has performative aspects that should be considered as such when studying these instruments. A thorough analysis of these narratives is therefore particularly enlightening to understand and possibly overcome the problems encountered in ES governance.

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Author Contributions

All authors contributed to the work presented in this paper. All authors read and approved the manuscript.

Conflicts of Interest

The authors declare no conflict of interest.

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