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"Limits of conventional cost-benefit analysis for the integrated management paradigm and the concept of ecosystem services" [1]

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“Limits of conventional cost-benefit analysis for the integrated management paradigm and the concept of ecosystem services”

Abstract

Cost-benefit analysis (CBA) represents for many economists a consensual referential approach allowing for the integration of economic evaluation into policy-making choices. However, implementation of CBA for the purpose of sustainability issues is confronted to real difficulties in evaluating ecosystem services. Although technical methods have been developed for solving these problems, a large critical literature of CBA still exists arguing its limits. This paper aims at building on these critics to show the failures of CBA in attempting to solve sustainability issues in the framework of the integrated management strategy. This strategy is consented at the highest international decision-making bodies to set societies on a sustainable path. This paper ends on two main results. i) First, we have identified 13 ecosystem services out of a list of 29 to which CBA should be exclusively restricted. ii) Second, we have argued why CBA should be exclusively located at later stages of decision making processes.

Keywords: cost-benefit analysis, sustainability, integrated management, holistic approaches, ecosystem services, willingness to pay, individual preferences, market failures, stated preferences, revealed preferences.

1. Introduction

Tools and methodologies for economic evaluation of the environment are helpful for decision makers in preserving or restoring environment quality at the least economic cost possible, as required by national and international regulations (e.g. European Commission's guidelines for the implementation of the water framework directive [18]). However, if not properly applied, economic methodologies may reveal unable to reduce environmental degradation. This is what happened during the last decades, management of environmental issues and conflicting anthropogenic uses have been framed by sector-related policies. As a result, numerous efforts have failed to improve environmental quality, as in European coastal zones for instance. According to the European Commission [11, p. 20], this is because environmental impacts were analyzed separately (i.e. through analytical approaches) whereas holistic[2] analyses were required [46, p. 25].

Indeed, decision-making processes are often supported by decision support tools such as conventional cost-benefit analysis (CBA). Conventional CBA and environmental economics in general encounter difficulties to take into account sustainability [34, p. 25] as well as physical limits to growth and strong sustainability [47], [34]. It results in high risk of underestimation of environmental benefits and low reliable figures when applied to issues that are too global or complex to be assessed in this manner. This is because those tools are based on low integration approaches and are too "micro-specific" (i.e. "analytic" as opposed

[2] By holistic, we mean any globalizing approach where various elements, usually isolated, are gathered and coordinated in order to achieve results more effectively. It relates to wholes or complete systems rather than analyses of, treatment of, or dissection into parts [32].

to “holistic”). The micro-specificity means that those tools are able to capture only a reduced number of impacts, territorial components and stakeholder antagonistic activities regarding the use of environmental assets. This is a major drawback since any environmental issue stems from antagonistic uses of environmental assets [8], [29], either in direct, indirect or induced effects. In addition, for the restricted number of environmental impacts that can be captured by analytical tools such as conventional CBA, their importance is likely to be underestimated.

In conventional CBA, these shortcomings (reduced number of captured impacts and underestimation of their importance) partly arise from the problem of the underlying concepts of monetary values based on individual preferences which are aggregated into one single indicator. This makes conventional CBA likely to underestimate environmental benefits, which in turn may distort the cost/benefit ratio to the detriment of benefits. Hence, it may influence decision makers in lowering or postponing environmental targets. This is for instance permitted by the European water framework directive when costs exceed benefits [18][3].

This paper builds on the CBA difficulties, which have been emphasized in economic literature for the last 30 years, to show the limits of conventional CBA to fulfill the integrated management paradigm[4]. Integrated management is considered at the highest

[3] Read the following pages: pp. 8-9, 12, 14-18, 24, 116-117, 123-127, 135, 196-197, 206, 208, 215-217.

[4] Detailed definitions of integrated management are available in [46], [11], [8], [38], [10], [33], [12].

international decision-making bodies as a possible strategy towards sustainability in the respects to the physical limits to growth imposed by the environment. Therefore, it is important to understand what can be the place of conventional CBA as a decision support tool in the process of integrated management. Through this aim, this paper is an attempt to answer two questions: “to which categories of ecosystem services conventional CBA should be restricted?” and “at which step conventional CBA should be located in the process of decision-making”?

This paper is structured as followed. In second section, a background on conventional CBA and the concept of ecosystem services are shortly presented. In third section, the main precepts of integrated management (considered as the sustainable reference) are presented. In fourth section, those precepts are confronted to the bases of the conventional CBA. In fifth section, recommendations for the use of conventional CBA are given.

2. General background on conventional cost-benefit analysis and the concept of ecosystem services

As a decision support tool used in environmental economics, conventional cost-benefit analysis (CBA) consists in a set of methodologies for economic evaluation of the environment. This set of microeconomic methodologies aims at quantifying the monetary value of changes (caused by a policy, a project, etc.) in the quality and/or the quantity of ecosystem services provided by environmental assets.

Under the term “conventional CBA”, are gathered two categories of methodologies. First, there are pricing techniques such as market methods, opportunity cost and replacement costs.

Second, there are revealed and stated preferences techniques. They respectively offer a surrogate and a constructed market solution and are justified by its users for goods and services suffering a lack of existing data on real costs and benefits, which is the case for most ecosystem services [27, p. 20]. Revealed preferences techniques include hedonic prices, travel cost method, recreational demand models, averting behavior and defensive expenditures models (also named cost-based valuation methods), cost of illness and lost outputs. Stated preferences technique includes contingent valuation and choice modeling experiment. A more detailed presentation of these methods is given in literature, *inter alia* [34].

In environment, conventional CBA is used to value the change in ecosystem services caused by a project or a policy. This value is obtained by aggregation of willingness to pay (WTP) of individuals for this change in a given territory (and for a defined period of time), exactly the same way it is happening in the framework of a market. WTP is the amount of money an individual would be willing to pay to secure a benefit or avoid a cost[5]. This amount is estimated through the set of methodologies mentioned above (pricing, stated and revealed preferences techniques). This concept allows for environmental measures to be ranked by people order of preferences. This is achieved by aggregating individual preferences into cost/benefit ratios. It allows then, projects with the highest benefits and the lowest cost to be selected.

This aggregation of costs and benefits conducts to the Total Economic Value (TEV) of an environmental asset. Environmental economists decompose the TEV into use and non-use

[5] Definition of benefits and costs can be found in [18, pp. 118-124].

values. "*Use values relate to actual use of the good in question (e.g. a visit to a national park), planned use (a visit planned in the future) or possible use* [option use value[6]]*"* [34, p. 86]. The first two categories (actual and planed use) may be categorized under direct and indirect use value. A direct use requires direct interaction between individuals and the environmental asset considered. An indirect use relates to ecosystem services used by individuals via the use of another ecosystem service (e.g. individuals that benefit from the natural detoxification of waste water by wetlands when they enjoy bathing in clean rivers). Non-use values relate to WTP to keep a good in existence in a context where the individual expressing the value has no actual or planned use for his/herself [34, p. 86]. Indeed, some people's willingness to pay for the conservation of an asset, independently of any use they make of it, is influenced by their own judgments about intrinsic value [34 , p. 19]. Non-use value can be categorized in terms of a) existence value, b) altruistic value, and c) bequest value. In existence value, motivations could vary and *might include having a feeling of concern for the asset itself (e.g. a threatened species) or a "stewardship" motive whereby the "valuer" feels some responsibility for the asset. Altruistic value might arise when the individual is concerned that the good in question should be available to others in the current generation. A bequest value is similar but the concern is that the next and future generations should have the option to make use of the good* [34, p. 86].

Use values may either concern market or non-market ecosystem goods and services – e.g. bathing in the sea or a river is a non market service since no entrance fees are required to

[6] Option use value relates to ecosystem services that people might use in future. Indeed, "*people may be willing to pay to maintain a good in existence in order to preserve the option of using it in the future"* [34].

enjoy it. Non-use values relate only to non-market goods – e.g. people that feel concerned by the existence of an insect species living in Antarctica because they believe that it has its own right to live or simply because they believe it has its importance in the worldwide ecosystem even if they do not know why and how (existence non-use value).

The second concept to be developed in this section is the concept of ecosystem services. This paper partly relies on the concept of ecosystem services because it enables the limits and biases of conventional CBA to be clearly identified. Indeed, it clarifies which components of environmental assets are effectively included in conventional CBA and which ones are inherently discarded due to intrinsic weakness of this approach. This is important since it is obvious that conventional CBA does not value an environmental asset in its whole but only some components of it, i.e. it values only few ecosystem services provided by this asset. This is maybe one of the reasons why the interest for the concept of ecosystem services has been growing exponentially since 1998 up to 2008 in scientific literature [20]. Fisher *et al.* [20, p. 645] give a general definition of ecosystem services: *"ecosystem services are the aspects of ecosystems [i.e. ecosystem structures, organizations, process and functions] utilized (actively or passively) to produce human well-being"*, either directly or indirectly. This is in line with the even broader definition from the Millennium Ecosystem Assessment [28, p. V]: *"Ecosystem services are the benefits people obtain from ecosystems"*. Ecosystem services can be classified into six categories as shown in table 1: provisioning, sink, supporting, regulating, cultural, and site [30], [28]. For example, water detoxification is an ecosystem service provided by environmental assets such as natural wetlands. A deeper discussion on these disputable definitions is given in Cordier *et al.* [13].

3. The integrated management paradigm

The practice of integrated management extends back at least to 1965 with the first integrated coastal management program by the San Francisco Bay Conservation and Development Commission. Then, it has been enforced at the United-states national scale with the US Coastal Zone Management Act in 1972 [44], [9], [39]. In the first decade, the practice was confined to the United States, Australia and the United Nation Environmental Program [44, p. 1-1]. But it progressively spread all over the world and in 1993, 75 countries and semi-sovereign states had initiated 217 integrated coastal zone management (ICZM) efforts at national and sub-national scales. At the beginning of 2002, those numbers respectively doubled and tripled amounting to 145 countries and semi-sovereign states that had initiated 622 ICZM efforts at national and sub-national scales [44, p.3-1]. *"Now ICZM is practiced all over the globe and is part of the rhetoric for sustainable development"* [44, p. 1-3]. Although the paradigm of integrated management has been profusely developed for coastal zones, it seems that its main precepts remain valid for any other kind of areas. This explains why the broader concept of integrated environmental management (IEM) has been developed in other countries, which is very similar in its definition to the ICZM paradigm [22, p. 20], [19].

In Europe, despite some early policies (e.g. the 1973 resolution on coastal areas), integrated management appeared only in the 1990's. It started first with the legally binding Convention for the Protection of the marine Environment of the North-East Atlantic (the « OSPAR Convention ») signed in 1992 and entered into force in 1998 for 15 western European countries. Second, it was followed by a comprehensive Demonstration Programme in the period 1996-1999 that included 35 demonstration projects located in European coastal zones [39], [41]. This program resulted in 3 communications and recommendations from the

European Commission, Council and Parliament [10], [33], [12]. Those documents, although not compulsory, set the basis for the implementation of integrated coastal zone management (ICZM) in European Union.

Integrated management consists in a sustainable strategy for an integrated approach to planning and management of anthropogenic activities. It encompasses the whole process of data collection, planning, decision making, and implementation management and follow-up. Its global objective aims at conciliating environmental quality targets with social and economic targets (i.e. sustainability). “Integration” means to unify parts together to make a whole. In Integrated environmental management, the term “integrated” corresponds to bringing together different components inside a single strategy [43]. In that strategy, all policies, economic sectors, administrative decision levels, territorial physical components and, to the highest possible extent, individual interests are taken into account and unified into a single strategy. Moreover, proper consideration is given to the full range of temporal and spatial scales. Such a strategy must involve all stakeholders in a participative way [38]. Attention is also given to the numerous tools required to achieve sustainability. They must all be integrated to the strategy [10], [33]. Hence our interest in the way conventional CBA may be integrated to the integrated management strategy.

Integrated management seems to offer a useful strategy to set society on a sustainable path. First, it contributes to fulfill the three pillars of sustainability: improvement in social, economic and environmental conditions. Second, it enables to take into consideration the physical limits to growth, an important problem highlighted by the European Commission [10]. Third, integrated management seems to incur quite low implementation costs but brings back high net benefits. An economic analysis carried out by European Commission [17, p.

38] supports this assertion. The study was based on the methodology suggested by Costanza *et al.* [14], [15], applied to 36 integrated management initiatives implemented in 13 countries mostly between 1995 and 1999. Although the methodology is highly disputable, it showed that each euro spent in the implementation of integrated management initiatives, brought back at least between 7,6 and 12,5 €$_{1999}$ net benefits (implementation costs have been subtracted). This range of values takes into account benefits produced by industries and tourism activities as well as non-market benefits provided by natural habitats.

4. Specific objectives of integrated management and failings of CBA

For its global objective to be achieved, integrated management targets five specific objectives allowing sustainability to take effect on field [11]. They will be individually analyzed in this section. The main failures of CBA related to those objectives will be argued based on a literature review.

4.1. The coordination between antagonistic uses

Sustainable issues are generally characterized by antagonisms and conflicts for the use of ecosystem services between stakeholders at different spatial and temporal scales [8], [29]. The integrated management strategy seeks coordination. Coordination is defined by Billé [8], as a way to reduce those antagonisms according to a logical plan. More precisely, it requires in a first step, identifying and managing antagonistic activities [29, p. 7], [8]. In a second step, inconveniences are distributed to stakeholders. Those inconveniences can take the form of compulsory cooperation between ministries, taxes, quotas, regulations, prohibitions, etc.

There are three categories of stakeholders' antagonistic interrelations regarding the use of ecosystem services [13]:

i) direct antagonism between stakeholders for the use of a same ecosystem service,
ii) indirect antagonism between stakeholders through the alteration of an environmental asset (but using two different ecosystem services provided by a same environmental asset),
iii) induced antagonism: the second-order impact of a change in one sector on another economically related sector caused by an environmental measure or an economic activity.

However, the quantitative description of the process leading to antagonism's causes and consequences is hardly possible in the conventional CBA framework. For instance, what happens in the situation where urban waste water treatment plant and metallurgy industries would see environmental targets made less stringent concerning river pollution by heavy metals. Losers would obviously be other companies that need to pump surface water for their industrial process. Their water treatment cost before use would increase with possible consequences for the competitiveness of that sector at an international level but also for another loser category: the employees that might lose their work due to restrictions caused by higher productions costs. This illustrates that CBA does not allow us to quantify what precisely happens to the losers (not only in monetary units but also in physical units) when winners are satisfied through a change in ecosystem services. This is because the final CBA output takes the form of one single aggregate resulting from the sum of individual WTPs. As a result, conventional CBA is technically able to take antagonistic interrelations into consideration only in terms of optimization of losses and benefits of different users.

The lack of antagonistic process analysis is striking in many CBA papers (e.g. [16], [24], [7], [23], [5], [37]). Interrelations between stakeholders are generally not even mentioned. Nevertheless, some conventional CBA papers do attempt to address interrelations between stakeholders. As an example, the study on the monetary value of wetlands conducted by Kontogianni *et al.* [23], explicitly mentions those kinds of conflicting interrelations. They address them through focus groups discussion between hotel owners, fishermen, farmers, etc. However, they are not integrated into their economic analysis nor quantitatively assessed. The study of these interrelations is restricted to the interpretation of the WTP components and the competing motivations for preserving wetland areas. This logic simply allows the different wetland preservation scenarios assessed by contingent valuation to be better interpreted. At least, it allows decision makers to be fully aware of the stakeholder groups they are going to favor and disadvantage. This is a partial attempt in solving the lack of contribution by conventional CBA to the need of coordination in integrated management. However this attempt is only partial because, the study of these antagonisms is neither integrated inside the conventional CBA nor quantitative. It takes the form of a qualitative social study that comes beside the economic analysis.

4.2. Holistic analysis against the micro-specificity of CBA – a critic based on the Millennium ecosystem assessment

Many experiences trying to improve environmental quality of different fields have failed in Europe, particularly in coastal zones. According to the European Commission [11, p. 20], these inefficiencies originate from the analytical approaches that have generally been favored. Contrary to past practices, holistic analyses are required [46, p. 25] and must include various kinds of (i) impacts expressed in monetary and physical units, (ii) stakeholders' categories and (iii) spatial and time scales at which impacts do occur.

Integrated management attempts to cover these three elements by carrying out holistic analyses of environmental issues. Analyses that consider a large number of ecosystem services will inherently conduct to cover a wide variety of impacts, stakeholder categories, spatial and time scales. This is first because environmental impacts result from antagonistic uses of ecosystem services by stakeholder activities (in direct, indirect or induced effects as explained above). Second, because ecosystem services are provided by environmental assets, which are located in diverse territorial compartments at various spatial scales (fish in waters, climate regulation in atmosphere, natural detoxification of waste water in soils, etc…). Third, because among the 6 ecosystem service categories (table 1), some occur at small time scales (e.g. cotton provisioning services) and other at mid- and long time scales (e.g. regulating services such as climate changes).

Moreover, in terms of decision-making efficiency, holistic approaches that focus on numerous ecosystem services decrease the risk of externality transfers. That can happen when policy measures are not assessed from a wide enough perspective. This potential risk does exist in the European Water Framework Directive. If the focus is only given to waters, water quality may improve while soil quality will be decreasing without even noticing it. Indeed, any urban or industrial waste water treatment rejects more or less clean water into rivers. However, a residual solid always remains after treatment in the form of sludge. When it is stored on land or spread on fields, soil quality is possibly at risk on the long term. If such a pollution transfer from one territorial compartment to another (from water to soils) is not clearly identified by a holistic analysis, no measure will be taken to reduce the risk of soil degradation on the long run.

Contrarily, conventional CBA is micro-specific (i.e. analytic) rather than holistic [1, p.6], [48, p. 550]. Conventional CBA assess a little piece of the world. It restricts the scope of issues to a micro-scale and leaves out global connections [45]. Indeed, conventional CBA needs the collection of such a huge number of data that it is time and money consuming. As a result of these technical constraints, economic analyses based on CBA methodologies focus only on very few specific issues or impacts [1, p.4]. For instance, CBA may assess the value of a change in natural wetlands surface through state preference techniques. However, it will neglect the role of this environmental asset in ecosystem services such as flood control, natural detoxification of water and soils pollutants, provision of habitats for biodiversity, etc. Although CBA's replacement cost techniques might offer an interesting complementary alternative, it still suffers drawbacks detailed below.

Given that ecosystem services are numerous and that as explained above, any environmental issue originates from conflicts for their use in direct, indirect or induced effect, conventional CBA micro-specificity turns out to be a huge drawback. Indeed, six main categories of ecosystem services can be listed as shown in table 1 [28], [30]: provisioning, sink, supporting, regulating, cultural, and site. They include at least 29 sub-categories of ecosystem services (e.g. illnesses-, climate- and flood-regulation, habitat creation for living beings). Nevertheless, most conventional CBA base the monetary value of their studied environmental asset on only one or two ecosystem services (e.g. recreational service provided by forest or wetlands). Limiting in such way the number of studied ecosystem services, leads to an underestimation of the total economic value of environmental assets and environmental impacts of some policies. Cuyno *et al.* [16] offer a good example of such micro-specificity. They assessed the environmental impact of agriculture. However, they have limited their CBA to the study of health effect of pesticides on humans and animals. As a consequence, the analysis was limited to the sole regulating services: disease regulation as well as pollination regulation through the effect of pesticides on beneficial insects that enable pollination (and hence crop production). More examples can be found in the following CBA papers: *inter alia* [7], [23], [5], [37].

Such holistic property based on a wide range of ecosystem services is impossible in conventional CBA because of its main underlying concept: individual preferences. Individual preferences cannot be counted on to fully reflect the importance of critical ecosystem services [34]. Indeed, conventional CBA fails to sufficiently encompass all services provided by environmental assets. This is due to the individual preference basis that exposes conventional CBA to the five following strong biases. Those five biases partly explain why "society" and the common well-being do not simply equal the sum of individuals' well-being although this is one of the major theoretical foundations of conventional CBA [34, pp. 16, 49].

i) Market failures: One of the main shortcomings of conventional CBA preventing sustainability to be fully taken into account stems from the calculation of environmental value based on individual preferences. The basic principle is very simple : the amount of money someone effectively pays when he buys any good, or would agree to pay if a market would exist for the concerned good, reveals the magnitude order of his preference : he would pay more for a good or a service he prefers. Such assertion assumes prices not to be influenced by market failures (e.g. oligopolistic markets, non internalization of negative externalities etc.). In other words, in conventional CBA methodologies, perfect market conditions are assumed. However, this is not often the case in reality, especially for critical natural resources [1, p.1]. As a result, market prices do not always reflect the real preference of individuals and pricing techniques based on real market data are strongly biased.

ii) Human cognitive limitations: Most individuals lack of knowledge and are not fully aware to weight up complex environmental issues with global effects occurring on a long period of time and/or large geographical scales[7] [27, p. 20], [29, p. 8], [4, p.132]. Yet, this is the case of regulating services, some sink services (e.g. CO_2 storage) and the most important ecosystem service that conditions the existence of all others: supporting services. The use of those three services is indirect (called "indirect actual use value" in conventional CBA vocabulary). People are not aware that they frequently use them and benefit from it via another ecosystem services. This is for instance the case of waste water detoxification in natural wetlands. This sink service is not directly used by individuals but they benefit from it due to less treatment costs needed for drinking water production and lower sale prices. It results in such a high uncertainty that no scientific knowledge can really quantify the expected impact of a change in indirect services provided by environmental assets. Therefore, it seems impossible that economics and individual preferences attempt to do so. This is confirmed by the Millennium ecosystem assessment [28, p. 101] that ranks supporting, sink, regulating services as well as cultural services as the most uncertain. Moreover, ecosystem services that people might use in future (option use value) are also very difficult to value by individuals since such future use is by definition unknown today.

[7] Large scale relates to intercontinental scales but also to national and regional scales as well as watersheds as small as 10 000 km^2.

This remains true not only for pricing techniques but also for surrogate market approaches. Indeed, even if information on ecosystem services are provided to respondents before starting the contingent valuation or to consumers before buying fish or any other environmental good on real market, they lack of knowledge to assess their economic monetary value. Regarding such problem, last column of table 1 is of particular interest. Indeed, according to the direct or indirect use of ecosystem services, their assessment through conventional CBA is respectively more or less reliable. Therefore, table 1 suggests that indirect ecosystem services should probably not be assessed through conventional CBA. As a result, only provisioning and site services as well as some cultural and a few sink services could be assessed through conventional CBA. This is a direct application of the concept of the *monetization frontier* developed by O'Connor and Steurer [29], [31]. According to this concept, ecosystem services are split into two groups: those that can be valued through monetization and those that cannot. In that framework, monetary units are only attributed to ecosystem services that can be valued in terms of their direct potential conversion into marketed goods & services (tables 1 and 2). Physical units are attributed to others (i.e. indirect ecosystem services).

Table 2 may help to clarify the principle of *monetization frontier* developed in table 1 for those who are familiar with conventional CBA and the concept of TEV. Indeed, table 2 matches the concepts of direct/indirect use of ecosystem services by individuals (as presented in last column in table 1) with conventional CBA concepts of use and non-use value as well as market and non-market environmental goods and services. If only direct individuals' use of ecosystem services should be valued through conventional CBA as suggested in table 1, table 2 leads to the conclusion that only use value of market goods and use value of some non market goods (those with direct use of ecosystem services by individuals) may be captured through conventional CBA. Other ecosystem services should be analyzed through complementary holistic and integrated approaches.

Table 2 shows additional information not addressed in table 1 about the TEV category of non-use values (definition given in section 2). It suggests that the valuation of non-use value in monetary units through conventional CBA should be restricted to the category of existence value. Indeed, existence value refers to personal and arbitrary value relating rather to own philosophical precepts than to any use of ecosystem service. Hence, there is no methodological shortcoming to allow an individual to set a monetary unit on a value created by his/herself as long as other use values and related ecosystem services are properly assessed in a way that fulfill the imperative conditions listed in section 5 and conclusion. On the contrary the two other categories of non-use value, altruistic and bequest values, relate to values for other people than the individual who expresses its WTP in the CBA. Indeed, the non-use category of altruistic value includes the idea that environmental assets should be available for others in the current time, for them to enjoy it either through existence value or through use value (i.e. by using ecosystem services). The bequest value is exactly the same as altruistic value except that the “others” that should enjoy the environmental asset are the next and future generations. As a result and by definition, altruistic and bequest categories of non-use values cover policy measures whose impact affects populations living in places remote in space or/and time (e.g. in other regions, other countries, on long time horizons) from the location of the individuals expressing their WTP. As seen above in section 4.2, such global space and time scales assessed through individual preferences are hardly likely to give reliable figures due to high complexity causing great uncertainty. This typically covers supporting, regulating and some sink services, which are not (directly) used by individuals (e.g. European individuals feeling strong concern for flood control in Pacific small southern islands affected by climate changes). As seen above, these ecosystem services are complex, highly chaotic and uncertain [3]. Therefore, they are really uneasy to be assessed through individual preferences methodologies like conventional CBA. As a result, altruistic and bequest categories of non-use values should probably not be specifically targeted by conventional CBA (although those two categories of non-use value are inherently included in individual preferences that are consciously or unconsciously expressed by people through real, constructed or surrogate markets).

iii) Salary limitation: In addition, in conventional CBA, stated preferences methodologies rely on interviews made to individuals where they are asked the amount they would be willing to pay for an environmental asset to be preserved or restored. However, individuals may underestimate the value of a change in the environment since they cannot afford to pay more than a certain percentage of their annual income [25], [21, p. 294]. As a consequence, any environmental measure that would provide benefits exceeding the value of this percentage would be underestimated. This comes to enforce underestimations caused by cognitive limitations.

iv) A partial solution does exist to cognitive and salary limitations: replacement cost (pricing technique) can cover some ecosystem services not addressed by constructed and surrogate market approaches such as stated and revealed preferences methods. This may concern some regulating ecosystem services (flood control, natural waste water treatments, etc.). Nevertheless, replacement cost methodology does not avoid classical conventional CBA drawbacks. Indeed, imperfection of market prices does not truly reflect the preferences of individuals expressed through the cost consumers are paying for technologies replacing destroyed natural services. In addition, individual preferences do not allow sustainability to be reached. First, because the market is coincidental so that it does not lead to an optimal use of environmental assets (read below). Second, because it might underestimate or ignore ecosystem services that cannot be restored and for which no replacement cost exists.

v) The assumed optimal use of environmental resources through market: despite observed market failures and the inability of individuals to really take global, long-time scale and indirect negative externalities into account (read above), the environmental economics tenants will argue that the market (real or surrogate), expressed through individual preferences in the form of WTP, leads spontaneously to an optimal use of (natural) resources. On the contrary, ecological economics tenants will argue that market does not always lead to a selection of optimal technologies, production activities, and use of space, and that even when prices are correct [47]. Therefore, ecological economics considers systems, including markets, as adaptive and coincidental rather than optimal. One important reason is that what exists today is not the result of the sole selection at the level of individuals. Inevitably, there is path dependence and lock-in, i.e. current economic changes highly depend on historical technologies [26]. And once the economy takes a certain technological path, positive network externalities appear so that users of a particular technology derive benefits from the simple fact that the number of other users increases. Such externalities arise because physical and informational networks become more valuable as they grow in size (e.g. hardware or phone networks). Then, as technologies are built one on another, little by little, the economy is locked-in in a path, whether this path is sustainable or not. In addition, lock-in processes do not only affect the environment, it may also be negative for the economic production itself (read *inter alia* [40], [36] and [26].

Table 1. Application of the *monetization frontier* to ecosystem services categories. Only direct use of ecosystem services by individuals should be valued into monetary units. Source: table modified from [28, pp. 7, 40-45, 50] and [30, p. 10].

Categories	Examples of sub-categories of ecosystem services	Direct / indirect uses by individuals *
SUPPORTING	**Services that are necessary to the production of the other 5 categories of ecosystem services presented below (regulating, sink, provisioning, cultural and site services):**	
	Life support for living beings (e.g. habitat creation, gene pool storage)	Indirect
	Nutrient cycling and soil formation required for food provisioning to plants, animals and human beings,	Indirect
	Primary production (plants and trees produced by photosynthesis) required for food provisioning services and habitat creation	Indirect
	Water cycling (conditions climate regulation, food production, etc.)	Indirect
REGULATING	**Services needed for regulation of ecosystem processes and reduction of natural disturbances under a level compatible with good quality life conditions:**	
	Climate regulation (and subsequently natural hazards such as hurricanes)	Indirect
	Erosion regulation	Indirect
	Flood regulation	Indirect
	Pollination regulation (e.g. regulation of pollinator insects populations)	Indirect
	Disease regulation (e.g. risk of malaria in southern Europe caused by climate changes)	Indirect
	Pest regulation (harmful for crops and livestock)	Indirect
SINK	**Natural capture, storage and detoxification of wastes:**	
	Water purification in natural wetlands	Indirect
	Pollutant filtration in clay soils before reaching underground waters	Indirect
	Natural sequestration of CO_2 (e.g. in oceans)	Indirect
	Landfills	Direct

PROVISIONING	**Products obtained from ecosystems:**	
	Food (crops, livestock, fish, games)	Direct
	Fresh water – see also water resources in supporting services)	Direct
	Wood and fibers (timber, cotton, hemp, silk) – see also wood resources in trees as an habitat in supporting services	Direct
	Non renewable resources: fossil fuels and material (e.g. mining products)	Direct
	Renewable energy from sun, wind, water, biomass	Direct
	Molecules for pharmaceutical purposes	Direct
	Genetic resources (e.g. genetic diversity is important to cross-breeding activities) – see also gene pool storage in the supporting category of services	Direct
CULTURAL	**Non-material services people obtain for ecosystems through spiritual enrichment, cognitive development, reflection, recreation and aesthetic experience:**	
	Cultural diversity (ecosystem diversity is one factor influencing culture)	Indirect
	Educational (ecosystems provide the basis for education in many societies)	Indirect
	Spiritual development	Indirect
	Recreational and touristic activities	Direct
	Aesthetic values	Direct
SITE	**Environment as a two or three dimensional space for economic activities:**	
	Infrastructures	Direct
	Mobility areas (e.g. fluvial and road traffic)	Direct
	Storage areas	Direct

* In the last column, “direct use” means that individuals use the ecosystem services directly and consciously. In the case of indirect uses, individuals use an ecosystem service via the use of another one (e.g. individuals unconsciously use the sink service of waste water detoxification in natural wetlands via the use of recreational services such as bathing in clean rivers). This distinction between direct and indirect uses is important to emphasize since we argue that only direct use of ecosystem services by individuals should be valued through conventional CBA. Most of the time, ecosystem services with direct use have a clear direct potential or actual conversion into marketed goods & services. Hence they can easily be valued through monetary units in CBA. This principle of *monetization frontier* (developed in [29] and [31]) remains quite restrictive and does not conduct to excessive monetization. This is because a same asset may provide at the same time direct services (monetizable in CBA) as well as indirect services (non-monetizable in CBA). For instance, as shown in table 1, it is the case of genetic resources that are categorized in provisioning services and also in supporting services.

Table 2. Categories of the Total Economic Value (TEV) that can be valued in monetary terms through conventional CBA.

<table>
<tr><th></th><th>Market goods and services</th><th>Non-market goods and services</th></tr>
<tr><td>Use value</td><td>MAY BE CAPTURED BY CBA
Direct use of ecosystem services by individuals
e.g. use of timbers sold on markets (provisioning services – table 1)</td><td>MAY BE CAPTURED BY CBA
Direct use of ecosystem services by individuals
e.g. bathing in the sea or in rivers (recreational service – table 1)
NO CBA
Indirect use of ecosystem services by individuals
e.g. indirect use (defined below table 1) of natural detoxification of urban waste water by wetlands (regulating service – table 1).</td></tr>
<tr><td>Non-use value</td><td>-
(by definition market goods and services with non-use values do not exist. Indeed, if a market is possible for a good, it means that people use it or could potentially use it)</td><td>NO CBA
Altruistic and bequest values given by individuals
e.g. European individuals feeling high concern for flood control (regulating service) in Pacific small southern islands caused by the worldwide policies of green house gas emissions (sink service)
MAY BE CAPTURED BY CBA
Existence value given by individuals
e.g. existence value: people that feel concerned by the existence of an insect species in Antarctica because they believe that it has its own right to live</td></tr>
</table>

Since it is argued in table 1 that only direct human uses of ecosystem services should be monetary valued, table 2 involves that conventional CBA could only be applied to:

- use values of market goods and services,
- some use value of non-market goods and services (i.e. those with <u>direct</u> human uses of ecosystem services only),
- only one category of non-use value: the existence value.

4.3. Economic and social integration

Holistic approaches enable integrated management not only to target natural ecosystems preservation but also to take account of economic and social considerations. This is essential since environmental and socio-economic goals are inherently linked to each other [11].

One single indicator such as the cost-benefit ratio used in conventional CBA is unable to describe reciprocal influences between social, economic and environmental components of anthropo-ecosystems. This is the same sort of problem as already explained in section 4.1 for antagonistic interrelation processes. Single indicators, monetary or not, do not enable interrelation processes between various components or parts of a whole to be studied (the components or the parts are here: social, economic and environmental conditions). Moreover, conventional CBA does not only suffer from a lack of integration of social, economic and environmental components. It is all the complexity of ecosystems, the dynamic and broad concept of sustainability that is impossible to encapsulate in one single indicator [42], [4, p. 137].

4.4. Stakeholders' participation:

Integrated management and the sustainability cannot be effective if stakeholders are not regularly included in a participative way [11], [46, p. 12]. This includes industries, local population, NGOs, decision makers, and any other stakeholder affected by the environmental issue taken into consideration. Promoting stakeholders' participation seeks two goals. i) First, building a collective regulation and minimizing contestation. Numerous examples in the European Union show that if stakeholders are not involved in decision making, they are likely to reject environmental planning drawn up by decision makers [11]. Indeed, a rule or a compulsory legislation without social legitimacy is unlikely to be respected by stakeholders. ii) Second, avoid environmental measures to be annihilated by unknown or misunderstood antagonistic activities. This requires gathering stakeholders for them to clearly identify and understand processes of antagonistic activities regarding environmental targets. Participation can take the form of various processes such as arbitration against some logics in favor of others, negotiation, dialogue aimed at reaching consensus, communication, awareness rising, unilateral decision taken by one decision maker, etc. Those processes end up at the creation of instruments enabling human activities to be regulated or "coordinated": taxes, laws and rules, agreements, norms, etc. [8].

However, conventional CBA does not seek stakeholders' participation to economic assessment. Instead, it is normative: it fixes *ex ante* the management criteria to adopt by ranking environmental projects from the lowest to the highest cost/benefit ratios and bypassing public discussion. Therefore, conventional CBA is suggesting to decision makers what to do instead of independently informing on consequences of each project. This makes governance possible without politics and is not compatible with dominant ideas of democracy [45] when used as the sole economic tool.

4.5. Uncertainty management:

Integrated management recognizes explicitly the future uncertainty and the precautionary principle [11]. This is a necessary condition for sustainable policies to succeed. Indeed, future problems are not easily predictable. Therefore, integrated management must be an iterative process and must be flexible in order to be regularly adapted to new issues.

As explained by Billé [8], the environment is complex and varies quickly in space and time. Therefore, it is impossible to achieve a complete understanding of natural systems. As a result, there is no complete scientific methodology enabling a full quantification of the importance of environmental assets for living being and human activities. This uncertainty and lack of complete scientific knowledge can drive to decision-making stasis. The lack of scientific knowledge is currently used to prevent any action in environmental preservation [8]. However, uncertainty is impossible to be completely removed and must therefore be managed because it is inherent to scientific approaches [35, p. 87], [1, p.6].

In conventional CBA, uncertainty originates from the fact that a huge number of ecosystem services cannot be valued. This is due to market failures, cognitive and salary limitations and market non-optimality as explained in section 4.2. It occurs especially for indirect actual use values and option use values. In addition, for ecosystem services that can be assessed, the magnitude of their social, economic and vital utility is likely to be underestimated. For instance, this uncertainty is very strong for environmental supporting and regulating services because they do not directly affect people. Nonetheless, they are the most vital ecosystem services because they are absolutely necessary for life and economic activities to be maintained on the long run. Indeed, supporting services affect humans only because they are necessary to the functioning of all other ecosystem services, which are themselves directly useful to anthropogenic activities and ensure human life [28, p. 40)[8]. Regulating services

[8] Examples of supporting services that provide provisioning and regulating services: i) fertile soil formation provides many provisioning services such as food products, ii) photosynthesis

affect humans because they ensure proper and safe conditions for all other anthropogenic activities to happen (without disease reducing available labor, floods destroying infrastructures, etc.).

As a result of ecosystem uncertainty faced by conventional CBA but also by all scientific knowledge, the precautionary principle is recommended in integrated management [3]. However, conventional CBA does not let enough room for uncertainty management. This is partly due to lack of holistic properties and stakeholders' participation. i) First, as seen in section 4.2, lack of holistic properties is inherent to conventional CBA techniques. This leads to ignore a vast range of important ecosystem services. As a consequence, important adverse impacts on anthropogenic activities may be omitted from the analysis. ii) Second, the absence of stakeholders' participation reduces uncertainty management in conventional CBA. Indeed, other techniques such as deliberative green accounting, lend easily themselves to participative social deliberation, which may help to manage uncertainty. In the *"interface flows deliberative green accounting approach"* developed by Cordier *et al*. [13], it is asserted that uncertainty requires a precautionary level of environmental minimum quality to be preserved. Then the way the economy can adapt to that level is analyzed. This level is collectively selected through social deliberation in a participative way. This choice is enlightened, but not set, by scientific knowledge since inherent ecosystems uncertainty does not enable us to know the exact sustainable level. Such deliberative green accounting comes to answer the question "how to adapt our economy to environmental requirements?",

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provides vegetal organic matter production allowing vegetal land cover to act as a regulating services (e.g. flood control by wetlands), etc. [28, p. 40].

whereas conventional CBA answers the opposite question “how ensuring environmental quality with current trend in anthropogenic activities”.

It is true that in conventional CBA, sensitivity analysis may help to see how the economic system varies according to parameters with highly uncertain value. But this can only be conducted for clearly identified and assessed ecosystem services. Yet, as seen above, very often, only one or two ecosystem services are assessed whereas more than 29 sub-categories do exist at least as shown in table 1.

5. Recommendations

For the integrated management precepts to be fulfilled and the concept of ecosystem services to be properly taken into consideration, conventional CBA should be used only if the 5 following conditions are satisfied:

5.1. The lack of holistic properties requires conventional CBA to be used in last steps of decision making processes. As explained in sections 4.2 and 4.5, the lack of holistic and participative properties makes conventional CBA to address a limited number of ecosystem services categories, i.e. a limited number of stakeholder categories and impacts. Therefore, it turns out to be valid and useful at microeconomic scale: scale of a project, an economic sector, an industry or a small group of individuals. Therefore, conventional CBA may illustrate and offer a specific focus on one or only few aspects of possible options of environmental measures already identified. Monetary value provided by conventional CBA becomes then one deliberation criteria among others relating to the options such as it is the case in multi-criteria analysis. However, at early stages of decision making, i.e. when public authorities have to identify and select a set of policies and measures for the purpose of their territory management, macroeconomic scale is more suitable than microeconomic scale. As a result, conventional CBA should not be used in first step of decision processes, i.e. in support to identification of possible political choices. In that case, important environmental, social and economic impacts of the options would be likely to be ignored.

5.2. We should probably restrict the use of conventional CBA to the valuation of direct use values only[9]. This is relevant given the high degree of uncertainty and underestimations in monetary values (section 4.2 and 4.5). Such restriction would limit the use of conventional CBA to provisioning, some direct sink services such as landfill, some direct cultural services such as recreational activities, and site ecosystem services (table 1). For instance, conventional CBA methodologies would be suitable for the valuation of the recreational service of coastal tourisms in the case of the assessment of projects aimed at decreasing oil spills (e.g. make legally binding the replacement of oil tankers every 10 or 20 years). In that example, recreational service is an ecosystem service with direct use by individuals. According to the rules set in table 1 and 2, this can be captured by monetary valuation through conventional CBA. It can be carried out through benefit losses for coastal hotels and restaurants due to damaged beaches and the subsequent decrease in tourism. In that case, conventional CBA would not have the pretention to assess the natural habitat offered by the beach to the coastal biodiversity, neither the flood regulation service, which are respectively supporting and regulating services (both are indirect individuals' use services as shown in table 1) offered by the coastal vegetation. Indeed, conventional CBA should not assess

[9] This condition concerns actual and planned use values. Option use values should be discarded from conventional CBA due to too high uncertainty as explained in section 4.2.

indirect use values (last column in table 1) such as supporting and regulating services as well as some indirect sink services such as waste water runoff treated in natural wetlands and some indirect cultural services such as educational and spiritual services. This is because they are not properly assessed through individual preferences due to the indirect characteristic of benefits they provide to individuals (section 4.2). It makes it more difficult to value by people, which are subjects to cognitive limitations (lack of awareness and knowledge). In addition, it is not only a question of cognitive limitation but also of complexity and resulting uncertainty (section 4.5). Indeed, causality links between indirect ecosystem services and impacts on human life and activities are often highly complex and chaotic. It results in such a high uncertainty that no one can reliably assess the expected impact of a change in those ecosystem services. This is confirmed by the Millennium Ecosystem Assessment [28, p. 101], which states that major uncertainties hindering decision-making process run for regulating, sink, cultural and supporting services. This uncertainty is even greater for option use values since it concerns future uses that are still unknown today (all 6 categories of ecosystem services can enter in the conventional CBA category of "option use value"). Due to the uncertainty inherently contained in the concepts of option use value, it should not be treated by conventional CBA. It should better be taken into account through uncertainty management by social deliberation with precautionary principle (e.g. some green accounting approaches [13]).

5.3. If conventional CBA methodologies are also applied to the valuation of services with indirect individuals' uses (in addition to the valuation of ecosystem services with direct use), they should perhaps be applied to market costs components only, not to benefits. Indeed, conventional CBA methodologies are of particular interest for instance in the case of floods due to climate change and lack of adaptation measures such as dams for instance.

Definitely, hedonic prices methodology reveals to be useful to assess the subsequent depreciation of real estate value. This information might be useful if land owners are to be compensated by public authorities for this loss (this is the case in the Scotland case study of Bateman *et al*. [6]). Actually, this condition 5.3 fulfills the condition 5.2. Indeed the market cost components of ecosystem services with indirect human use may be envisaged under another perspective and categorized into other categories with direct use. For instance, in the example mentioned above, the market component of the flood regulation service (indirect individuals' use as shown in table 1) is the ecosystem service of safe site (direct individuals' use as shown in table 1) available for human infrastructures and habitations. Another possible alternative could be to consider this market component inside the category of provisioning service (direct individuals' use), i.e. as "safe lands free of flooding" provided by the ecosystem for human infrastructures and habitations.

5.4. Altruistic and bequest categories of non-use values should be specifically targeted by conventional CBA? Probably no. As seen in section 4.2, maybe the valuation of non-use value in monetary units through conventional CBA should be restricted to the category of existence value. On the contrary the two other categories of non-use value, altruistic and bequest values, cover policy measures whose impact affects populations living in places remote in space or/and time from the location of the individuals expressing their WTP. As seen in section 4.2, such global space and time scales assessed through individual preferences are unlikely to give reliable figures due to high complexity causing great uncertainty. Therefore, altruistic and bequest categories of non-use values should probably not be specifically targeted by conventional CBA if condition 5.2 is to be respected.

5.5. Holistic and integrative approaches should be developed and used at early stages of decision-making process. They would allow for the identification and selection of policy measures [13]. This is the corollary of condition 5.1 that excludes conventional CBA from early stages. Those holistic and integrative approaches must cover the 6 ecosystem service categories, both direct and indirect services. Supporting services should especially not be excluded from the analysis because they are the most important and vital services: all other ecosystem services as well as life continuation are conditioned by supporting services (table 1). The lack of holistic and integrative properties conducts to underestimations of ecosystem services importance. This lack contributes to maintain a vision of sustainable development in which sustainability is located at the interconnections of three circles (social, economic, environment). However, the integrated management vision of sustainable development considers those three circles as concentric (from inside toward outside circles): economic development is constraint by social development, which is constraint by environmental physical limits. Those theoretical physical limits are nevertheless difficult to translate into environmental norms and figures. Therefore social deliberation is required [13]. As asserted by Ackerman [1, p.5], environmental laws and regulations in United-States between the 1960's and 1970's have been extremely successful at reducing pollution and protecting health and nature; "*although adopted, for the most part, without complex economic calculations, none of these protective measures have bankrupted us or proved unaffordable*" (e.g. first 80% lead removal from gasoline in 1970's [2]). Social deliberation between politics was the main decision support tool, e.g. the Federal Clean Air Act [2, p. 12], [1, p.6].

6. Conclusion

This paper suggests that when used as the sole economic tool, cost-benefit analysis cannot be considered as an efficient approach for supporting decision-making in sustainable issues. However, in spite of its limits largely pointed in the literature, it remains an interesting tool for the purpose of integrated management as a complementary support of economic information which can be included into a wider multi-criteria analysis. Nevertheless, our paper advocates that conventional CBA would better take sustainability into consideration and fulfill the requirements of the integrated management strategy if the 3 following conditions were simultaneously respected.

1. If conventional CBA is used in the selection of policy measures, it should be applied only in last steps of decision-making, after the set of possible policy measures or projects have passed through a screening step that discards those that do not ensure sustainability in the respect of the environmental limit to growth. Such screening step might for instance occur through a deliberative process with stakeholders' participation at early stages of decision-making process as suggested by Cordier *et al.* [13].

2. Conventional CBA should be applied only to ecosystem services with direct use by individuals (tables 1 and 2) – this covers actual and planned direct use values but excludes option values. If expressed in CBA terms, this second rule restricts the use of conventional CBA only to use value of market goods and use value of some non market goods (those with direct use of ecosystem services by individuals). Concerning option use value, it should be assessed with tools that are more capable to cope with high degree of uncertainty and to consider the precautionary principle. Additionally, non-use value of existence can be covered by conventional CBA whereas altruistic and bequest values should be discarded.

3. If beside the valuation of direct ecosystem services, as advocated in condition n°2, conventional CBA methodologies are also applied to the valuation of ecosystem services with indirect human use, it should be applied to market costs components only, not to benefits (e.g. financial compensation of land owners in the case of depreciation of real estate values caused by floods due to the lack of adaptation measures to global warming). Note that this 3rd condition is exactly equivalent to the 2nd condition except that it is presented under another perspective (read section 5.3).

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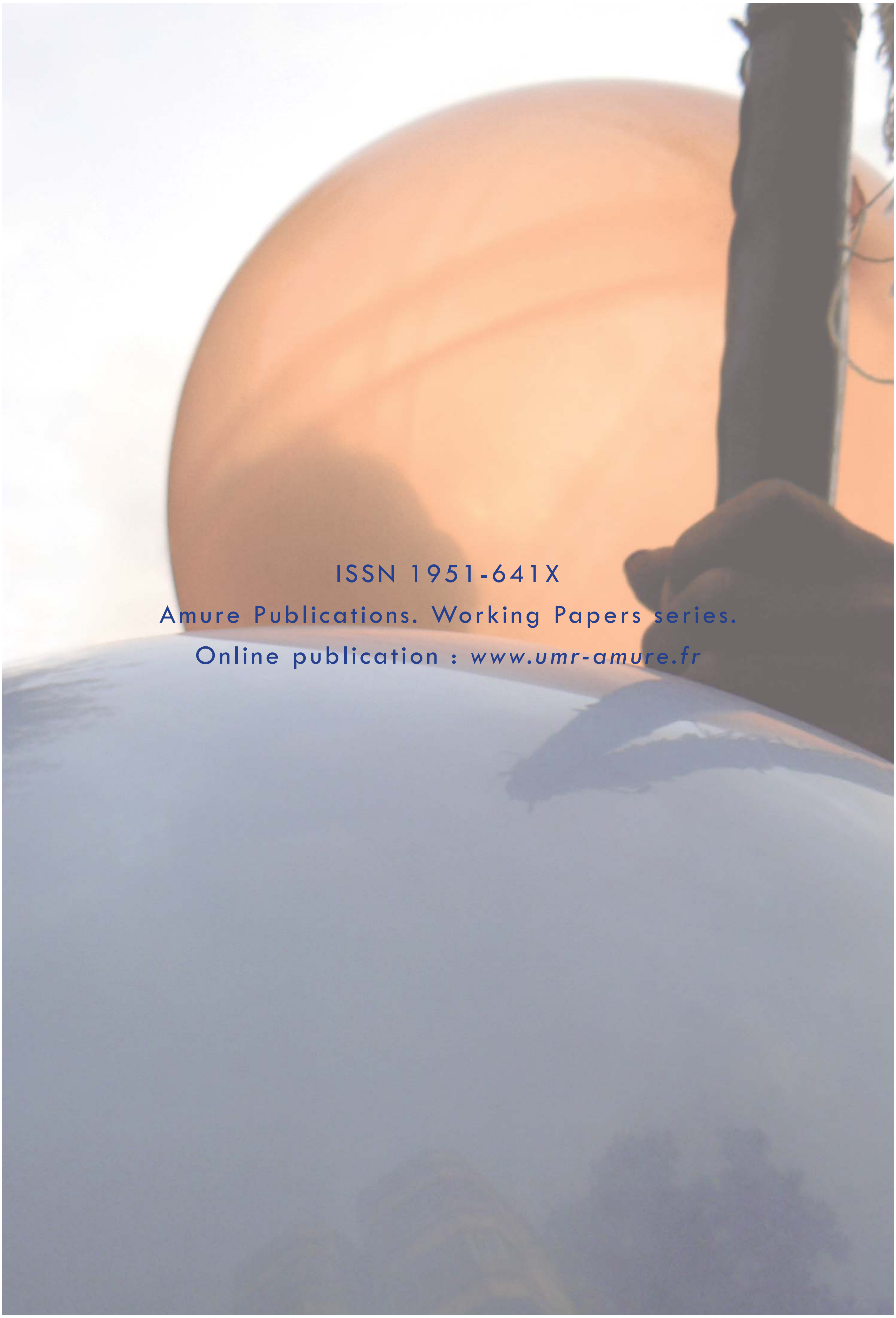

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