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Investigating the inclusion of ecosystem services in biodiversity offsetting

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ABSTRACT

In response to growing international interest regarding the consideration of ecosystem services (ES) in the framework of biodiversity offsetting (BO) and the current lack of guidelines on the subject, we investigated the potential inclusion of ES in BO, highlighting the risks and opportunities. Our argument is premised on the assumption that a practical link already exists between the two and that most of the tools required to make this approach operational are available. But so far, ES are not explicitly taken into account when calculating and designing offsets (whether regulatory or voluntary). One way to integrate ES in BO is to use the Environmental Impact Assessments' framework, here we propose a logical way to integrate ES at each step of the implementation of the mitigation hierarchy and provide details on the links with existing practice. In our proposal, the inclusion of ES is presented as a way to complement current approaches based on the assessment of habitats/species/ecological functions rather than to replace them. We argue that measures proposed to offset biodiversity losses, in addition to respecting ecological performance standards, should equally be chosen to minimize residual losses of ES. The latter require offsetting by different types of complementary measures. Implementing these recommendations as good practice should strengthen the weight of biodiversity, demonstrate consideration of social equity, and result in better acceptance of development projects and the measures proposed to offset them.

1. Introduction

Biodiversity offsetting (BO) is increasingly used in environmental policy as a way of reconciling economic development and the conservation of biodiversity; its objective is to achieve No Net Loss in biodiversity. The aim of BO is to counterbalance the negative impacts on biodiversity arising from development projects by providing ecological gains through conservation or restoration actions. Offsetting is the last step in the mitigation hierarchy, which aims first at avoiding, then reducing, and finally offsetting residual impacts on biodiversity. While BO requirements are not new (they have appeared in the environmental regulations of many countries over the last four decades), the concept has recently benefited from renewed political interest and has been endorsed in various policies, such as those of the Convention on Biological Diversity (CDB) and in the biodiversity strategies of a number of member states in the European Union (EU).

The concept of ecosystem services (ES), defined as the benefits that humans derive from nature emerged at the end of the 1970s in the scientific arena.[2] ES differ from the concept of function defined as the fundamental ecological structures and processes but also as the potential that ecosystems have to deliver a service (Braat and de Groot, 2012). ES original aim was to raise awareness,

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[2] Ecosystem services are commonly divided into four categories. Provisioning services describe the material or energy outputs from ecosystems (food, water and other resources), regulating ones act as regulators (regulating the quality of air and soil or by providing flood and disease control), supporting ones are necessary for the maintenance of all other ecosystem services (e.g. biomass production, production of atmospheric oxygen, soil formation and retention, nutrient cycling, water cycling, and provisioning of habitat) and cultural ones are nonmaterial benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experience (MEA, 2005).

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particularly of politicians, concerning the value of biodiversity and the costs of its degradation (Norgaard, 2010). But since the Millennium Ecosystem Assessment (MEA) (2005), which "firmly placed the concept of ecosystem services on the policy agenda" (Gómez-Baggethun et al., 2010), ES has been extended to the scientific community and become a significant focus of research dealing with biodiversity issues (Méral, 2012).

Conceptually, the principle of offsetting according to the goal of No Net Loss can be applied differently depending on what is at stake – habitat/species, ecosystem functions or ecosystem services (Calvet et al., 2015a; Levrel et al., 2012a). Currently, most offset practices focus on habitats and species, but are increasingly integrating a functional approach. Indeed, current methodologies to size offsets rely on five key features: the definition of detailed target components of biodiversity and ecosystems, the selection of appropriate indicators often based on an area calculation, the identification of appropriate baselines for calculating losses and gains, time-related issues and uncertainties in both assessment and offset outcomes (Quétier and Lavorel, 2011). Many authors have stressed that current BO relies mainly on a biophysical approach (e.g. Mann, 2015; McKenney and Kiesecker, 2009; Quétier and Lavorel, 2011). Others have highlighted a lack of consideration of social and cultural aspects in BO implementation, which may be a source of injustice and inequality (Apostolopoulou and Adams, 2015; Burylo et al., 2013; de Billy et al., 2015; Gobert, 2015). Indeed, the location where offsets are put in place does not necessarily provide ecosystem services to those who have lost them at the location where the impact occurred (BenDor and Brozovic, 2007; Gobert, 2015; Landsberg et al., 2013; Ruhl and Salzman, 2006). This question is of particular significance when the subsistence of a population relies on the ecosystem services impacted by a project (Sullivan and Hannis, 2015). Acknowledging the importance of this issue, the international community has begun in recent years to call for the consideration of ES in BO programs (Ives and Bekessy, 2015). Yet ES offsetting lacks a framework to facilitate its implementation (Bidaud et al., 2015), which is likely the result of significant knowledge gaps concerning this new approach that has only recently been included in policies (Braat and de Groot, 2012; CBD and UNEP-WCMC, 2012; CSBI, 2015). It is also affected by a great deal of debate between scientists on the use of this concept in conservation strategies (e.g. Schröter et al., 2014).

In light of both the renewed interest in this subject and the lack of guidelines regarding it, we conducted an investigation of the potential use of ES in BO, highlighting the risks and opportunities. We propose a conceptual framework of ES inclusion in BO that emphasizes the consequences on current practices. Our work was based on the premise that practical links between BO and ES already exist and that most of the tools required to make this approach operational are currently available. As observed by Duke (2014), "because biodiversity is a key element of natural capital, many of the conventional instruments by which we seek in practice to conserve it […] also serve, even if they were not explicitly designed to do so, to safeguard natural capital and ecosystem services". In our study, offsets are discussed within the regulatory and voluntary contexts of anticipated and accidental impacts, although the scope for accidental situations is quite limited. It deals solely with BO – not with carbon offsets, which can be considered as compensation focused on only one ES. It should be kept in mind that any discussion of BO is necessarily embedded within the broader context of the mitigation hierarchy.

The paper is organized into three parts. Section 2 investigates the current inclusion of ES in biodiversity offsetting both in regulatory and voluntary contexts, in academic literature, and in other unexpected contexts. Section 3 highlights the potential benefits and limitations of including an ES approach in biodiversity offsetting, and Section 4 proposes a framework for defining authorized impacts in which consideration of ES complements the mitigation hierarchy as it is currently implemented and offers a more thorough way to ensure the achievement of biodiversity conservation goals.

2. Existing links between ES and offsetting: where things stand

2.1. Regulatory contexts

The Convention on Biological Diversity's (CBD) 2011–2020 strategic plan on biodiversity, including the Aïchi objectives signed at the 10th Conference of the Parties in Nagoya, Japan, gives guidelines on how to support largescale actions for biodiversity conservation. These guidelines do not provide any details regarding the mitigation hierarchy or the implementation of BO, nor do they mention a potential link between BO and ES. Nevertheless, the plan commits the 168 signatory parties to developing national strategies for biodiversity, in which one tool is mitigation.

Action 7, Target 2 of the EU Biodiversity Strategy to 2020 (European Commission, 2011) aims to achieve 'No Net Loss' of ecosystems and ecosystem services through measures that include the development of offsetting schemes. Concerning regulatory frameworks, historic regulations related to the implementation of the mitigation hierarchy (e.g. the Environmental Impact Assessment (EIA) Directive [85/337/EEC] and its amendments, the Habitats Directive [92/43/EEC] and the Water Framework Directive [2000/60/EC]) do not mention ES. However, regarding accidental impacts, the Environmental Liability Directive (2004/35/EC) recommends service–service and resource–resource approaches for sizing offsets. The REMEDE working group[3] recommends using two specific ecological-equivalence scaling methods: Habitat Equivalency Analysis (HEA) for the service–service approach and Resource Equivalency Analysis (REA) for the resource–resource approach. HEA is commonly used in the United States. However, it should be noted that the term 'service' is related to 'functions' in these methods.

At national level, we decided to detail the French case study as France is the only European country with Germany to fully mandatory require offsetting for certain biodiversity impacts (Conway et al., 2013; Tucker et al., 2014). France's Biodiversity Strategy does not mention the concept of ES in relation to BO. French policy related to the mitigation hierarchy (dating to 2012) considers the need to take ES into account, but remains vague. Guidelines on the implementation of the mitigation hierarchy (MEDDE, 2013) list the different areas that must be considered to ensure equivalence between losses and gains, and suggest that ES could be considered under the ecological aspects, which are regarded as a priority area. In addition, two other areas, geographical/functional and societal, call upon similar concepts to those of ES, without mentioning it explicitly. In 2014, a new French law to protect biodiversity was drafted, however, it is still being reviewed and amended in parliament and is not expected to pass before the end of 2016. Whether or not to include the concept of ES within the mitigation hierarchy is still under debate. While a previous version of the legal text called for the avoidance and reduction of impacts on both biodiversity and ecosystem services, but required offsetting only for biodiversity, the last available version of the legal text (January 2016) more generally calls for avoidance, reduction and offsetting of impacts on environment (without mentioning ES).

In the United States, the implementation of BO has a longer

[3] Resource equivalency methods for assessing environmental damage in the EU (REMEDE).

history than in Europe, and the approaches developed there can be considered an attempt to grasp the value of certain ecosystem services. In terms of anticipated impacts, ES are mentioned in the introduction of the Final Rule of the 2008 Compensatory Mitigation for Losses of Aquatic Resources (USACE and USEPA, 2008), but are not taken up in the articles of the act. The concept of ES is used to justify the need to regulate impacts on wetlands and other habitats, however, the impact assessment methods outlined usually do not refer to ES. Yet some authors, such as Robertson et al. (2004), have described wetland mitigation banking as “a market in privately owned ‘wetland ecosystem services’, such as duck habitat, flood protection and biodiversity, seen as a way of achieving the goals of the US Clean Water Act of 1977 (CWA)”. Indeed, mitigation bankers are paid by developers for restoring habitats and ecological functions: in other words, promoting ecosystem services. This interpretation of mitigation banking implies that all impacted wetlands provide, more or less directly, ecosystem services to humans. In terms of accidental impacts, Habitat Equivalency Analysis (HEA) was created by the US National Oceanic and Atmospheric Administration (NOAA) in 1995 and incorporated into the Natural Resource Damage Assessment (NRDA) process (Dunford et al., 2004). This assessment process was then included in the Oil Pollution Act (OPA, 1990) and the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA or Superfund, 1980), the objectives of which are to offset environmental damage caused by oil spills or chemical pollution on land and/or in a coastal zone. As mentioned in Bas et al. (2016), HEA uses a single indicator to express the level of ecological function or service lost or gained. In this case, although the term ‘ecosystem services’ is used, a closer look at the meaning of ‘service’ indicates that it refers more to ecological functions.

In the different regulations we analyzed, we did not find any explicit operational recommendations on including the concept of ES within the framework of BO.

2.2. Voluntary contexts: international standards in the private sector

Beyond the legal requirements regarding the mitigation hierarchy and biodiversity offsetting, some voluntary initiatives to improve or standardize the implementation of these tools are currently emerging from the private sector. These include standards and guidance issued by the International Finance Corporation (IFC), the Business and Biodiversity Offsets Programme (BBOP), and the Cross-Sector Biodiversity Initiative (CSBI).

The IFC helps its clients to manage the environmental and social risks of their projects through the publication of Environmental and Social Performance Standards (PS) and Guidance Notes (GN). Projects funded by the IFC have to comply with IFC Performance Standard 6 on Biodiversity Conservation and Sustainable Management of Living Natural Resources, wherever the project is located, including in countries where nature conservation is poorly regulated. This standard requires the application of the mitigation hierarchy to impacts on biodiversity and ES (paragraph 7 of PS6, 2012b), and thus to offsets. Ecosystem services considered priority must be offset in potential cases “where socioeconomic and cultural uses of biodiversity (i.e. ecosystem services) are at issue” (point 31 of GN6, IFC, 2012a). Two types of priority ES are considered: those likely to be impacted by the project's operations, resulting in negative impacts on affected communities; and those on which the project’s operations depend (paragraph 24 and 25 of PS6, IFC, 2012b). For the former, offsets may include the provision of ‘compensation packages’ for affected communities (as outlined in several other IFC PSs), with a preference for collective in-kind compensation.

The IFC itself is a member of another well-known initiative, the BBOP, an international collaboration between companies, institutions and organizations, both private and public, to develop best practice in BO (GN32 of Guidance Note 6, IFC, 2012a). The 2012 BBOP Standard on Biodiversity Offsets makes a link with IFC PS6 (BBOP, 2012 p.4), explaining the relationship between BO and ES in its introduction and "as biodiversity underpins ecosystem services, the focus of the Standard is on ensuring no net loss of biodiversity, but there are important links to ecosystem function and services" (BBOP, 2012 p.15). However, the BBOP's ‘Principles, Criteria and Indicators’ do not explicitly include a way to apply offsets in terms of ES.

The stakeholders of many current activities that have impacts on biodiversity generally recognize IFC PS6 on biodiversity conservation. For instance, the International Petroleum Industry Environmental Conservation Association (IPIECA), the International Council on Mining and Metals (ICMM), and the Equator Principles Association launched the Cross-Sector Biodiversity Initiative (CSBI) in early 2013 to “develop and share good practices and practical tools” in applying IFC PS 6. A recently published CSBI report on implementing the mitigation hierarchy mentions some limitations such as the fact that "Equivalence (whether or not the option represents fair and appropriate redress) may be an issue where potential offset sites are substantially different from the impact site(s). Where offset and impact sites are far apart, loss of ecosystem services for particular stakeholder groups may also be a consideration—this can be a significant social risk for certain projects." (CSBI, 2015 p.67).

For example, in the case of Ambatovy nickel and cobalt mining enterprise located in Madagascar, the BBOP report (Berner et al., 2009) mentions that a cost-benefit model and analysis was applied to determine the ecosystems services to the local communities in and around Ankerana as well as the mine site. Ambatovy adheres to environmental standards including the Equator Principles, the IFC Performance Standards, and the principles of the BBOP.

In the same line, other grey literature from sources such as the World Resources Institute (WRI) (Landsberg et al., 2013), the Secretariat of the Convention on Biological Diversity, and the United Nations Environment Programme's World Conservation Monitoring Centre (CBD and UNEP-WCMC, 2012) have suggested general implementation recommendations for using ES in the mitigation hierarchy and in BO. For instance, the WRI proposes six steps for conducting an ES review throughout the environmental and social impact assessments scoping, baseline and impact analysis, and mitigation stages, including Excel spreadsheets. In their Viva case study, a mining project in the Arctic that would include an open mine pit, a processing plant, a port, a slurry pipeline, and a new access road, they identify measures to mitigate the loss in wild condition of traditional hunting areas experienced by hunters such as scheduling project related road transportation. They also proposed restoring the hunting areas as part of project decommissioning and closure. To offset residual loss in satisfaction with their hunting experience, affected hunters would be permitted to use the project's private roads in order to expand their range and access new hunting grounds.

So international standards developed by the private sector provide the first elements in how to put ES to use in the framework of BO by requiring that offsets for ES must be implemented in addition to those for biodiversity, though these are limited to what are considered as priority ES. The offsets should be sized on the basis of livelihood and well-being and be implemented collectively (rather than focusing on individual interests) and, whenever possible, in kind. But the use of cash compensation or justifying the absence of compensation when it is not technically or economically feasible is not excluded.

2.3. Some examples of the inclusion of ES within BO from academic literature

Academic literature on biodiversity offsetting is increasingly abundant, but examples of including ES in sizing offsets are very scarce even today. Calvet et al. (2015b) counted 477 articles (dating from 1984 to 2014) on the topic of biodiversity offsetting in a detailed bibliometric search on the Web of Science database. Special issues of scientific journals addressing this topic are also growing in number (e.g. Devictor, 2015 – special issue of *Biological Conservation*; and Froger and Hrabanski, 2015 – special issue of *Ecosystem Services*). However, when a search for the key words 'ecosystem services' and 'offset' was performed on the Web of Science database, it revealed far fewer articles (192), all published since 2004. Of these, around 30% take into account only ES related to carbon dioxide regulation, and only around 10% are strictly related to ES and offsetting. The majority of the articles use the term 'ecosystem services' as synonymous with function or biodiversity.

Many researchers have drawn attention to the lack of focus on ES, such as Tallis et al. (2015), who highlight "the need to move away from area- and habitat-based assessment methods for both biodiversity and ecosystem services [and] towards functional assessments at landscape or seascape scales". Wainger and Mazzotta (2011) specifically call for interdisciplinary science research on the subject.

One study including ES was done by Wende et al. (2015), who present a case study related to a project in Berlin to replace a velodrome with a shopping centre. The authors consider that Germany, by including ordinary biodiversity and abiotic parameters to the assessment of impacts and their offsets, has a suitable framework to apply the mitigation hierarchy to ES. In other words, so far, ES have not been systematically and explicitly included in offsetting. Two types of equivalence assessments were carried out in this example. The first one is based on functions and also includes urban ecosystem services such as climate regulation (the cooling capacity of the microclimate, air movement, and dust/pollutant scavenging capacity), soil sealing (the ratio of sealed surface to flow), and aesthetic considerations.). The habitats were assessed in terms of units corresponding to a score (reflecting the quality of the ecological functions and ecosystem services provision, the method is not detailed in the article) multiplied by the surface of the habitat and then mapped. The second one is based on biodiversity replacement costs. Some of the ecosystem services such as soil sealing were thus monetized.

In another study, Arbelaez and Sagre (2015) discuss an example of a proposed compensation scheme for the Cerrejón open-cast coal mine in Colombia. The aim of the developer was to demonstrate the equivalence between the ecosystem services lost at the impacted site and the ones gained through the implementation of the biodiversity offsets (Carbones del Cerrejón Limited y Conservación Internacional, 2012). The possible offsets involved measures related to reforestation, ecological restoration, funding of a protected area extension, management plans for fauna and flora of interest, etc. However, the authors do not specify the assessment methodologies used for these various types of offsets.

In a study on voluntary BO in the context of two mining projects in Madagascar, Bidaud et al. (2015) explain that mining companies have not yet integrated ES in offsets' calculation (in-kind and financial offsets). The companies use methodologies based on the habitat hectare metric. However, they try to incorporate ES in their strategies. The first company undertook an economic valuation of the ecosystem services provided by the offset sites (Olsen et al., 2011). The second company "tries to quantify the degradation of ecosystem services in order to restore the functionality of those services to their users", but no further details on the methodology are mentioned by the authors.

A recent study by Mandle et al. (2015) emphasized the need to carefully choose the spatial location of BO in order to avoid social inequality. They tested their method of tracking changes in ES benefits on a road construction project in the Peruvian Amazon. Using the software InVEST[4] (Integrated Valuation of Ecosystem Services and Trade-offs), based on spatially explicit, ecological production-function-based ES models, they assessed four different ES: sediment retention, nitrogen regulation, and phosphorus regulation (for surface drinking-water quality) as well as carbon storage (for climate regulation). They concluded that taking into account an approach based on a 'service-shed' rather than solely on 'ecological processes-shed' would reduce the unmitigated impacts, such as to drinking water quality.

Though the examples presented here are not exhaustive, they represent a range of recent practices related to the consideration of ES in biodiversity offsetting; albeit some are only remotely linked to the approach put forward in our study.

2.4. Implicit inclusion of ES within BO within environmental impact assessments

In fact, even if it is not frequently explicitly mentioned, the link between ES and BO has implicitly existed for a long time, as Aronson and Moreno-Mateos et al. (2015) found for the relationship between ES and ecological restoration.

However, the tools for sizing and implementing ES offsetting is another question. Environmental Impact Assessments (EIA) are internationally recognized as an essential tool in implementing the mitigation hierarchy in the context of development projects (although some countries still do not require their implementation). For this reason, the role of EIAs as "suitable tools to mainstream information about ES in decision-making" (Geneletti, 2013a) has already drawn much attention (see Geneletti (2013b) – special issue of *Environmental Impact Assessment Review*). Baker et al. (2013) mentioned that an ES approach needed to improve the EIA process, and in particular the "environmental outcomes that it delivers". In practice, ES are very rarely mentioned in EIAs, as noted by Tardieu et al. (2015) in the specific context of linear infrastructures. In our literature review, we noticed that ES are already included in current EIA practice without being explicitly mentioned, although their inclusion is incomplete. Environmental Impact Assessments rely on analyzing impacts on and risks to physical, biological and socioeconomic environments. By explicitly considering ES in EIAs, the links between the impacts on physical and biological environments and on socioeconomic environments could be made more visible, contributing to moving away from the current silo-based approach. The structure of an EIA is well suited to integrating ES in the framework of BO, provided that the risk of double-counting ES is avoided (see Section 3.2.2).

In some cases, biodiversity offsets already consist of ES-oriented measures, compensating more for the costs to humans than for the costs to biodiversity No Net Loss goal. For instance, in recent marine and coastal projects, the creation of artificial reefs in the vicinity of the project is often proposed as a biodiversity offset, but in fact its aim is to offset both biodiversity losses and fishery losses. Although there is debate on the contribution of artificial reefs in attracting or producing reef fish (e.g. Boehlert and Gill, 2010; Inger et al., 2009), this measure can be regarded as a good example of an offset for the provisioning service of fisheries. Levrel et al. (2012b) reinforced this observation through a case study in Florida showing that the indicators used to assess ecological equivalence were based on limiting social conflict; the aim of offsets was more to compensate divers and fishermen than to mitigate ecological losses. One example is the use of boulder reefs (a type of artificial reef), which favor an abundance of big fish (providing cultural services), but do not compensate for the ecological impacts of projects or accidents (for instance, by providing

[4] www.naturalcapitalproject.org/InVEST.html.

regulation services such as playing the role of a nursery).

Besides mandatory biodiversity offsets, which are required for any significant residual impacts on the environment, other types of measures can be proposed voluntarily by project developers in the framework of an EIA or through a negotiation process outside the EIA. These are part of a broader category of measures called 'community benefits'[5] in the United Kingdom (Bristow et al., 2012; Walker et al., 2014) or 'accompanying measures'[6] in France (MEDDE, 2013). Their aim is to provide compensation of some kind for people affected by the negative environmental impacts of a project and/or to improve the social acceptance of the project (Kermagoret et al., 2015). In some cases, these may be related to different types of ecosystem services. As they are not considered biodiversity offsets, they are not sized according to the ecological losses caused by the project and do not necessarily involve in-kind actions, as is the case for monetary compensation payments. For example, some measures compensate local people for changes caused by the project to the environment that could impact their quality of life or safety. In a study by Kermagoret et al. (2014, p.12) on a planned offshore wind farm in the bay of Saint-Brieuc in France, scallop reseeding measures were proposed to compensate fishermen for scallop beds made unsuitable for exploitation by the construction of the wind farm. We would argue that this can be seen as compensation for a provisioning service. Another French project, the Dunkirk LNG natural gas terminal, has proposed accompanying 'community measures' based on the fact that people will no longer be able to access a beach used by anglers, hunters, kite boarders, windsurfers, walkers and birdwatchers. The proposal to create a nature centre and a natural bathing area, safety equipment for kiteboarding, a recreational lake near the terminal and to authorize access to the terminal under certain conditions for scientists to observe animal species mainly offset cultural services.

Other measures, resulting from negotiations with the developers and not specifically linked to the impacts of the project, can also concern improvements to ecosystem services. In the case of the Saint-Brieuc offshore wind farm, examples of this are the installation of chilled tanks for lobsters to improve a provisioning service and funding for projects to control the common slipper shell (*Crepidula fornicate*), an invasive benthic species to improve a regulating service.

These examples demonstrate, with more or less conclusive results, that the concept of ES is already integrated into some offsetting contexts, even if it is not explicitly mentioned as such.

3. Challenges and opportunities in integrating ecosystem services in biodiversity offsets

3.1. Benefits of including ES in the offsetting process

3.1.1. A broader definition of the environment

Current offset practice is mainly focused on safeguarding remarkable biodiversity (usually protected species and their habitats) rather than on ordinary biodiversity and its functions. Introducing ES in offset practices may allow non-scarce natural constituents that provide ecosystem services to be taken into account in the same way as more remarkable biodiversity (Baker et al., 2013; Burylo et al., 2013; CBD and UNEP-WCMC, 2012). Semi-natural and human-exploited environments could also be considered (CBD and UNEP-WCMC, 2012). The maintenance of pollination services, very relevant to the agricultural sector, relies strongly on the management of land cover.

3.1.2. Integration of socioeconomic and societal issues

Biodiversity offsetting rarely considers human populations who suffer from environmental losses generated by development projects and those that benefit from offset actions, regardless of the level of dependency of local communities on ecosystem services in maintaining their livelihood. Including ES in BO proposals may help to link human activities and amenities to affected or restored ecosystems (Lucas, 2014; Jax et al., 2013; Baker et al., 2013), making the offsets more fair and ethical. Through this approach, better justice should be achieved as ES beneficiaries are identified both at impact and offset sites. Besides the educational value of discussing ES with stakeholders during the consultation process, this could also be helpful in determining the ES and natural features to which individuals are attached (Baker et al., 2013; Schröter et al., 2014). Lastly, taking ES into account in BO could help improve the acceptance of projects, or even the acceptance of offsets themselves, which can be a key part of the negotiations concerning a project.

3.1.3. The consideration of indirect and cumulative impacts

The inclusion of ES in assessments may incite developers to consider the indirect or cumulative impacts of their projects by evaluating if "the project contributes to existing and foreseeable drivers of ecosystem change" and highlighting "whether and how a project could interact with ecosystem changes external to the project" (Landsberg et al., 2013). For instance, people may suffer 'indirectly' from a decrease in water quality miles away from the destruction of a wetland.

3.2. Limitations and questions related to including ES in the offsetting process

3.2.1. A controversial method for biodiversity conservation

The consideration of ES in biodiversity conservation is a highly debated topic among academics. Some argue that 'ecosystem services' are a socially constructed concept used to support a recent trend in nature conservation that promotes a utilitarian view of biodiversity rather than a scientifically grounded concept (Barnaud and Antona, 2014; Laurans et al., 2013). One of their concerns is that natural features not considered to provide 'services' would be excluded (Maris, 2014). For some, ES represents a tool that facilitates the commodification of nature and involves a dramatic narrowing of the views of biodiversity and the values attributed to it (Robertson, 2004; Mann, 2015; Maris, 2014). In this perspective, the concept of ES is not neutral and may reflect a specific view of conservation governance that obliterates the plurality of approaches within this realm.

The very idea of biodiversity offsets that would provide both sufficient ecological gains and ecosystem services gains can sometimes appear incompatible. Some ecosystems may supply more 'utility' when their ecological status is poorer: such as a wetland or stream, for which paths and other access points could be built. This contradiction is likely to be more problematic for provisioning and cultural ecosystem services than for regulating services, as mentioned in the CSBI (2015, p.13) where "increasing access to, or use of, productive services (such as wood fuel or fisheries) could be incompatible with improved biodiversity conservation, and with some regulating or cultural services". Another important concern is that some empirical studies have shown that

[5] Community benefits are defined as "some form of additional, positive provisions for the area and people affected by major development"(Bristow et al., 2012).

[6] Accompanying measures can consist of knowledge acquisition, the definition of a broader conservation strategy, the implementation of a biotope protection order overseen by national, regional or local governments, etc. They can be defined to improve the efficacy of or give additional safeguards to the environmental success of offset measures. They can also target socioeconomic activities.

an increase in ES does not necessarily lead to an increase in the level of biodiversity, and that uncertainties remain about the arguments that protecting ES prevents the erosion of biodiversity (Bullock et al., 2011; Harrison et al., 2014; Palmer and Filoso, 2009). For instance, increases in afforestation could be associated with an average water yield reduction or atmospheric regulation could be reduced in grassland communities due to increased mortality of root and rhizome tissues from grazing (Harrison et al., 2014).

3.2.2. Remaining methodological gaps

Currently, there is no standardized definition of the concept of ES, leading to multiple lists of these services (e.g. MEA, 2005; TEEB, 2010; MAES 2013), although there have been attempts to try to develop a stable definition (Munns et al., 2015). Equally, there is no unified framework or methodology for the biophysical assessment of ES (Tallis et al., 2015), nor are there standard indicators for assessing it. Liquete et al. (2013) describe the different existing typologies as follows. Capacity indicators are based on potential ecosystem services that depend on biodiversity, which is close to a habitat/species/function approach (e.g. fish abundance per site for the ecosystem service of food provision). In contrast, flux indicators are based on human use of the capacity (e.g. fish catch (kg/year) for the ecosystem service of food provision). Benefits indicators are based on a monetary value resulting from the economic exploitation of the flux (e.g. financial income from fisheries (USD/ha/year) for the ecosystem service of food provision). In this way, we notice that the assessment of losses and gains through a habitats/species/function approach implicitly contains an assessment of the losses and gains of ecosystem services, as biodiversity components are the basis of ES supply. Incidentally, when developing an ES approach, attention should be paid to avoid double-counting.

Difficulties encountered in offset sizing based on a habitat/species/function approach would similarly apply to an approach considering ES (Bull et al., 2013; Gonçalves et al., 2015) such as the determination of requirements for demonstrating no net loss of biodiversity, the characterization of threshold regarding biodiversity values beyond which offsets are not acceptable, the management of uncertainties throughout the offset process. Among other things, ecological trajectories of ecosystems are still pending questions. Biodiversity gains should be calculated in view of a counterfactual scenario that does not overestimate biodiversity loss without offset (Maron et al., 2015). Offset design is particularly intricate in a context where ecosystem dynamics are affected by climate change (Doswald et al., 2012).

Other impediments can be mentioned. When implementing the mitigation hierarchy, a quantitative assessment of ES losses and gains requires data that can be difficult to obtain, in particular because of shortcomings in understanding the relationship between biodiversity, functions and ES (Braat and de Groot, 2012; Brownlie et al., 2012; Cardinale et al., 2012; Schröter et al., 2014).

However, some tools try to incorporate ES in offsetting, such as OPAL[7] (Offset Portfolio Analyzer and Locator), open-source software that enables users “to estimate the impacts of development activities on terrestrial ecosystems and several of the services they provide, and then to select offsets to efficiently mitigate losses”, as explained in the OPAL user's guide.[8] OPAL creates static maps based on the InVEST nutrient, sediment and carbon models. Static maps related to other types of ES can be created independently. According to the CSBI (2015), InVEST and ARIES[9] (Artificial Intelligence for Ecosystem Services), another tool for identifying and prioritizing ES, may improve the definition of current baselines and trends, as well as potential project impacts. These tools allow assessments to better take into account variability over time and space of ecological and socioeconomic conditions linked with ES, as well as the demands of people losing these ecosystem services (Tallis et al., 2015). But limitations remain such as data availability and the lack of knowledge regarding the relationships between services and their potential proxies.

3.2.3. A plurality of values and social preferences

The concept of ES is deeply intertwined with the notion of value. Acknowledging the fact that it is impossible to quantify all the different social values of nature (Ives and Bekessy, 2015; Moreno-Mateos et al., 2015), it is crucial to be aware of the “dangers that some uses of the concept have in obscuring certain types of value”, as mentioned by Jax et al. (2013). These values are subjective and vary according to geographic, human and temporal contexts (e.g. Cáceres et al., 2015). For instance, a species that is common now at a large geographical scale might be rare decades later: in this case, the value assigned by society to this species is likely to increase with time. An ES approach is therefore often criticized from an ethical point of view because the concept has both a descriptive and prescriptive dimension (Jax et al., 2013).

‘Ecosystem disservices’, defined as the negative impacts of some natural elements on human well-being (i.e. the opposite of ES), are less studied (Sandbrook and Burgess, 2015). Some biodiversity offsets may be seen as generating disservices to humans living near the compensation site (e.g. a wetland that attracts insects). In fact, the same ecosystem can provide both services and disservices (Moreno-Mateos et al., 2015) but individual perceptions about these may vary. Vaissière et al. (2014) discussed the example of mudflats and marine worms, which may be considered as dirty or disgusting by some vacationers, while naturalists may be passionate about these marine habitats and the flora and fauna they host.

Thus when considering ES in offsetting, a key factor that should be taken into account is the identification of all the values associated with the concerned ecosystem services, as well as who the beneficiaries are (Baker et al., 2013; Jax et al., 2013).

3.2.4. Risks related to weakening equivalence

If ES were used alone in offsetting – that is, replacing rather than complementing a habitat/species/function offsetting approach – some abuses could be anticipated (however, it should be noted that a strictly ES approach is not the current trend). In this case, issues related to the notion of equivalence would arise.

One reason for this is that the integration of ES in offsetting expands the possibilities of achieving equivalence. So if only ES was considered, since several species/habitats may deliver the same ES, substitution between very different habitats would be allowed.

Secondly, it could facilitate the substitution of natural capital by human-made capital; for example, an impact on the ecosystem service of water purification delivered by a wetland could be offset by the construction of a sewage plant. This type of situation could also be favored by the inappropriate use of economic valuation which could make these two elements commensurable.[10] It might also lead to weak sustainability measures being accepted, such as increasing public attendance in animal parks to offset an impact on wildlife observed by birdwatchers (Hay, 2015). Hence, there

[7] http://www.naturalcapitalproject.org/software/#opal.

[8] http://ncp-dev.stanford.edu/~dataportal/opal-releases/1.0.0/OPAL%20User%20Guide%20v%201.0.pdf.

[9] www.ariesonline.org.

[10] Commensurability relates to the idea that different types of value can be expressed in a common measurement unit (Neurath, 1925, 2005; Kapp, 1965, 1983; O'Neill, 1993) in Gomez-Baggethun et al. (2010).

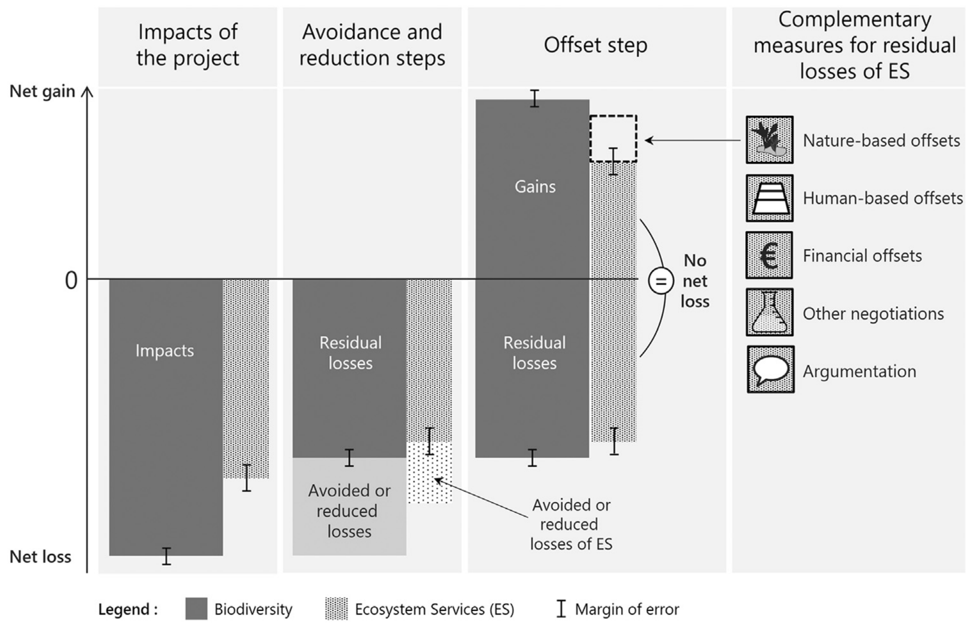

Fig. 1. Ecosystem services assessed alongside biodiversity using the mitigation hierarchy.

would be a risk of exacerbating a sectoral approach if certain stakeholders strongly lobbied to maximize the type of service they benefit from, in particular regarding provision services (e.g. food, raw material, water).

Thirdly, equivalence could be sought in a strictly monetary approach in which the impacted population directly receives cash payments from developers. Indeed, Gomez-Baggethun et al. (2010) show that the use of ES approach facilitates the commodification process through monetary valuation.

3.3. *Weighing up the pros and cons: ES as a complementary approach*

The overall objective of the mitigation hierarchy and biodiversity offsetting, as stated in current legislation, is No Net Loss of biodiversity. Acknowledging the different limitations we have discussed, it would be hazardous to determine BO considering only the services an ecosystem provides. ES cannot be considered as an accurate proxy of biodiversity. However, it can provide supplementary information that could be regarded as useful when assessing the impact of development projects on the environment as a whole (including physical, biological and socioeconomic aspects).

In a context in which the temptation is great to simplify current offsetting methodology, which is seen as complicated, it seems important to argue that integrating ES in the current approaches in a complementary way that respects their different objectives and rationales is crucial for better biodiversity outcomes. A solely ES-based approach cannot replace current approaches based on habitat/species/function assessment, an observation also made by the BBOP (2012). The latter are crucial in determining impacts on the biological environment. Nevertheless, an assessment of ES provides additional information on how impacts on the biological sphere will, in turn, impact the socioeconomic sphere. It might contribute to rebalancing impact accounting and determining trade-offs, giving more weight to biodiversity. We argue that considering ES should be complementary to the current assessment approach, and carried out as a second stage (the first stage should focus on guaranteeing ecological equivalence between the offsets and the project's residual ecological impacts). This is similar to the idea that an ES approach can be implemented alongside policies preserving biodiversity without replacing them, as recommended by many scientists (Faith, 2012; Reyers et al., 2012).

4. A way forward

We have discussed, in Section 2.4, that current EIA practice already includes ES in its assessment, but that this is done in an incomplete way. To build on this, we propose a logical process for integrating ES as an aspect to be considered for mitigation within Environmental Impact Assessments, linking this with existing practice. As biodiversity is the foundation that allows the provision of ES, our first premise is that "part of a loss in ecosystem service benefit might be mitigated by implementing the mitigation hierarchy on environmental impacts" (Landsberg et al., 2013). It should be noted that the following applies only to *anti*cipated impacts and not to accidental impacts. Fig. 1 shows how ES could be assessed alongside biodiversity using the current environmental mitigation hierarchy.[11]

4.1. *Initial state and impact assessment*

The first step is an initial assessment of biodiversity and ES at the site where impacts are foreseen; this is necessary in order to evaluate how much biodiversity and ES will be lost. As is currently the case for biodiversity, a project should be reviewed or abandoned if potential unacceptable ES losses are identified that cannot be offset, as stated by Brownlie et al. (2012) and Landsberg et al. (2013). This is the step in which the project is considered in the light of needs/objectives, environmental stakes and alternative solutions and a choice is made whether or not to move forward with the project. This is particularly important for priority ecosystem services, especially in developing countries, where certain residual impacts can jeopardize the survival of some populations.

[11] At each step, a margin of error reminds us that our knowledge and capacity to recreate nature limits our ability to guarantee that No Net Loss of biodiversity is achieved. Regulating services (Jessop et al., 2015), biodiversity and ecosystem functionality (Moreno-Mateos et al., 2012) are likely not fully recovered. This margin of error equally exists for ES losses and gains.

4.2. Choosing biodiversity offsets that maximize ecosystem services' supply

If a project is maintained, the impact on biodiversity is also likely to lead to an impact on ES (Fig. 1). The next step is to use the avoidance and reduction steps of the mitigation hierarchy to minimize losses of both biodiversity and ES. When choosing offsets, the ecosystem services supplied should be taken into consideration. This is particularly important for priority ES. Populations impacted by the project should be involved both in the process of identifying the ES provided by the site as well as those impacted by the project; offsets must compensate the impacted populations. If there is a choice between two ecologically equivalent offsets, the one that best compensates the impacted populations, namely in terms of ES, should be selected (e.g. restoration of the wetland that best limits flooding near the potentially impacted population). Thus, BO alternatives are first selected according to a functional approach and then the most suitable BO is determined depending on ES approach. However, finding a suitable compensation site to attain No Net Loss of biodiversity is already a challenge in itself, so this step may be impossible to implement if only one suitable offset can be proposed. Also, sometimes ES cannot be offset by ecosystem-based approaches; for example, the spiritual values of a natural area can be considered as unique. Regarding the choice of indicators to use to monitor ES, at this step it is preferable to use capacity indicators, but it is also possible to use indicators of flux or benefit because in any case the target is No Net Loss of biodiversity.

4.3. Complementary offsets for residual losses of priority ES

Given the aforementioned limitations (see Section 4.2), the design of offsets achieving both NNL of biodiversity and NNL of ES is currently challenging. Even if the No Net Loss of biodiversity is reached, residual losses of ES will necessarily remain (the empty dotted square in Fig. 1) and will not be offset through the regular process. From a socio-economic point of view, it is preferable to continue the process toward a No Net Loss of ES, we thus propose an approach where complementary offsets would be developed to address these residual losses of ES.[12]

There are different types of complementary offsets for ES residual losses (implying an evaluation of ES losses and gains) that could be implemented:

1. *Nature-based complementary measures* compensate with natural capital. In other words, they are measures based on ecological restoration or other actions that restore nature, such as biodiversity offsets. For example, the "creation of community woodlots to compensate for restricted access of local communities to forests due to the project" (Landsberg et al., 2013, p.42). The CSBI (2015, p.67) encourages 'composite offsets' that are a combination of "an offset at a landscape level and another offset closer to the impact for local affected communities". Capacity indicators should be best adapted to sizing nature-based measures.
2. *Human-based complementary measures* substitute natural capital with human-made capital. An example would be a "wastewater treatment facility to substitute for converted wetland" or "pharmaceutical medicine to substitute for disease control by undisturbed forests" (Landsberg et al., 2013, p.42). Indicators of flux should be best adapted to sizing human-based measures.
3. *Financial complementary measures* compensate populations that have lost ecosystem services with cash: for example, "for residual income loss from impacted fisheries" (Landsberg et al., 2013, p. 42). This type of offset is often criticized as it can be considered a way to 'buy people off'. Other more complex financial measures aim to artificially improve the provision of ES from a natural asset. Landsberg et al. (2013, p.42) give the example of a measure consisting of "investment in plant to process coffee so that the income per kilo of coffee increases". It should be noted that the more offsets are based on financial measures, the higher the risk of individual rather than collective offsets. Indicators of benefits should be best adapted to sizing financial measures.

If the residual losses of ES are considered acceptable (for instance, if no priority ES are impacted), developers may decide not to implement any of the complementary offsets mentioned above, and propose other types of offsets. These may result in *negotiations* until a compromise has been reached. In this case, the indicator is the acceptance of the stakeholders involved in the negotiation, so there is not necessarily either an evaluation of ES losses/gains or the achievement of equivalence. There are a number of different types of offsets, more or less related to the impacted environment, that may emerge from negotiations; these may include the financing of research programs, public awareness-raising or education campaigns, or other measures associated with local deadlocks (one example is the production of a documentary film on nature and marine reserves in the North Sea in the framework of the Compensation Plan for the Egmond aan Zee offshore windfarm[13]). We do not consider these negotiated 'custom-made' offsets problematic since other offsets deal with No Net Loss of biodiversity. However, they may be risky for impacted populations in the case of opportunistic developers, a lack of knowledge of stakeholders of their dependence on ES, or strong lobbying from certain stakeholders. Some developers may try to hide or minimize the ES residual losses as they sometimes do with ecological residual losses (Vaissière et al., 2014).

It is finally important to note that, as it is possible that very poor populations might prefer short-term financial compensation to long-term nature-based or human-based measures, or may not even be aware of the loss of ecosystem services they will suffer, it is important that the assessment of the initial state of ES is based not only on local community knowledge but on external scientific contribution.

4.4. Acceptable residual losses of ES that are not offset

Some residual losses of ES (all or part of the empty dotted square in Fig. 1) may be considered as acceptable and be resolved through *argumentation* related to the public interest of the project: socioeconomic (such as an increase in employment) or urgent need (such as a hospital, a road that makes a village accessible, or an education centre). This argumentation must be carefully prepared to attain acceptance of the project. Some developers manage to avoid having to implement complementary offsets by cleverly presenting their project and cultivating their relationship with the population to be impacted. However, the option of not offsetting residual loss based on convincing argumentation should only be used for non-priority ES.

At the end of the EIA process, no unacceptable losses of biodiversity or ES that are not offset should remain as these should

[12] Some people can also benefit from net gains of ES when BO is implemented in a place where people do not suffer from losses related to the project. If complementary measures are proposed to offset residual losses of ES caused by the project, this can be considered as social gains at a higher geographical or system level (except if this BO generates 'disservices').

[13] http://www.noordzeewind.nl/en/project-en/compensation-plan/compensation-plan/.

have been identified during the assessment of the site's initial state and during impact assessment (see Section 4.1).

5. Conclusion

The inclusion of ecosystem services in EIAs in order to offset impacts to these has received growing interest in the international community in recent years. Although ES are not explicitly included in current regulations and seem to be rarely mentioned in existing practice, our analysis shows that, especially in the EIA process, they are already implicitly considered, albeit in an incomplete way. Weighing up the pros and cons of including ES in offsetting, we conclude that focusing on ES in a second phase, after the goal of No Net Loss of biodiversity has been fully considered, is important. We propose a conceptual framework for integrating ES as part of the EIA process throughout Section 4. If these good practices are followed, it should give weight to biodiversity (since this underlies the provision of ES), as well as strengthen the consideration of social equity and result in better acceptance of projects and proposed offsets.

Currently, offset practices aim at achieving No Net Loss of certain aspects of biodiversity, but not overall biodiversity. As BO is required only when residual impacts are considered 'significant', aside from offsetting impacts on wetlands with the aim of restoring both function and species, most offset measures target protected species through the ecological restoration of their habitats. Indeed, 'impact significance' is a fuzzy term – in Europe, it is (relatively) clearly defined only in the procedures related to the Habitats Directive and the Water Framework Directive. In projects not subject to these directives, EIAs hardly ever identify significant residual impacts and hence offsets. Thus if offsetting practices remain the same, our hypothesis on the supply of ecosystem services by biodiversity offsets would be jeopardized. Our proposal relies on moving towards a better integration of ecological functions targeted by offsetting. Brownlie et al. (2012) state that "impact assessment alone cannot resolve global challenges of biodiversity loss and deterioration of ecosystem services that underpin human well-being; these issues must be dealt with at a strategic political level".

We have described a case-by-case approach to BO, but a macro-level approach could also be worth exploring. Various countries have developed a broader and more integrated vision of offsetting (e.g. Jacob et al., 2015; Quétier et al., 2014; Vaissière and Levrel, 2015), including initiatives such as mitigation banking in the United States or compensation pools in Germany. These demonstrate that it is possible to create ecological restoration projects in advance in priority areas to offset identified ES. These large-scale restoration projects aim at offsetting several development projects; these 'global' offsets are potentially more effective than individual trade-offs and have a better chance of ecological success (GAO, 2005). They avoid the temporal losses that often affect biodiversity and impacted populations (Bull et al., 2013). Indeed, although biodiversity offsetting is theoretically supposed to be carried out prior to impacts, it is very common that case-by-case offsets are implemented during or after a project. Duke (2015) states that ES could be made operational via BO and habitat banking. Concretely, this would imply choosing available credits in an offset that fits the need of the concerned ES at the step described in Section 4.2. However, because these pooled offsets have not been created ad hoc for a specific impact, they may not exactly supply the required ecosystem service.

To conclude, in the light of the legal vacuum and the near absence of guidance on the inclusion of ES in offsetting, the different options we present as complementary measures can be envisioned. There remain important issues to address: for example, how to assess equivalence and trade-offs between different levels of equivalence. The current ambiguity around the integration of ES within offsetting could become detrimental, favoring opportunism, inequality and injustice.

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