Biodiversity offsets and payments for environmental services: clarifying the family ties

Keywords

Ecosystems, biodiversity offsets, payment for environmental services, conservation policies, economic incentives

Abstract

Biodiversity Offsets (BO) and Payments for Environmental Services (PES) are sometimes used interchangeably to characterize innovative economic tools to conserve or restore biodiversity, ecosystems, or their services. We assume that a confusion between PES and BO can have negative implications for biodiversity conservation. In this paper, we argue that these two tools follow different targets and have different founding principles, and thus, their basic mode of functioning would only coincide under special circumstances and institutional contexts. Here, we propose a new definition of BO, delimiting them more clearly from PES, and use practical examples to underscore conceptual differences. Both tools require specific policy framework conditions, in terms of rights, responsibilities, and enforcement. If unmet, however, the implications for biodiversity conservation outcomes are stronger for BO than for PES since BO are explicitly linked to biodiversity losses, while PES typically are not. PES experiences can certainly inform BO implementation vis-à-vis contract design and enforcement, but these PES lessons need to be enacted vis-à-vis BO specific requirements, in order not to underestimate generic risks in their implementation: if a PES scheme fails, payments can be stopped; if a BO fails, biodiversity losses remain.

1 Introduction

Biodiversity Offsets[1] (BO) and Payments for Environmental Services[2] (PES) are both innovative tools to address environmental problems. The literature on their design is rapidly expanding (e.g. Calvet et al. 2015; Ezzine-de-Blas et al. 2016; OECD 2013; Wunder et al. 2018). Some authors have tentatively distinguished BO from PES (e.g. Hahn et al. 2015; Maseyk et al. 2016), but the two are typically seen as intertwined, and often as generically identical instruments. For instance, Milder et al. (2010) see BO as a potential PES to alleviate rural poverty in developing countries and Salzman et al. (2018) describe BO as "biodiversity PES programmes" (excluding several cases that they do consider as provision of a specific service). The widely referenced standard on BO developed by the Business and Biodiversity Offsets Programme (BBOP) (2012) states that 'one potential mechanism for securing the conservation outcomes needed for a biodiversity offset is payments for ecosystem services'. The BBOP adds that 'a range of people and organizations, from indigenous peoples and local communities, to farmers, NGOs, local authorities and protected area management boards, can be paid to deliver the specific conservation outcomes needed for the biodiversity offset to achieve no net loss (or a net gain)'. This standard conveys the progressive interest or need of including social aspects, such as poverty alleviation, equity or stakeholders' participation, to biodiversity conservation objectives.

From a regulatory perspective, some jurisdictions use terms to describe policies or instruments where BO and PES are being mixed together. For instance, in China, 'eco-compensation' is described as a "combination of 'ecological compensation' and 'payments for ecosystem services' " (Shang et al. 2018: 162). In France, parliamentarians recently recommended developing PES to pay farmers to deliver BO for developers, although PES are not mentioned in the country's Environmental Code as an option for BO implementation. Both the National Assembly (Bassire and Tuffnell 2018) and the Senate (Dantec 2017) argue that PES can help resolve some of the challenges of BO implementation in France, following the recent strengthening of No Net Loss requirements (Vaissière et al. 2018a), and in particular the acceptability of BO to farmers and farmland owners (Calvet et al. 2019).

The confusion between PES and BO may be due to similarities in structure, such as a conditional payment for providing environmental services, and the shared reference to "market-based instruments" in some policy circles (e.g. Convention on Biological Diversity), as discussed by Boisvert et al. (2013), Gómez-Baggethun and Muradian (2015), Pirard (2012) and Vatn (2015).

In practice, it thus seems that BO are often included into a "large-umbrella" PES family and market-based instruments. The need to improve the implementation of BO could lead to a hasty designation of BO as PES. However, as we will argue, BO and PES exhibit some quite distinct features. BO aim to counteract biodiversity losses from economic development by requiring developers to offset their impacts elsewhere through 'measurable conservation gains' (i.e. transforming financial payments into biophysical performance). PES are aimed at incentivizing/rewarding changes in land or resource-use practices, to deliver improved environmental outcomes, including biodiversity conservation, typically to a set of users willing to pay for incremental environmental services. The underlying rationale and

[1] We use BO to refer indiscriminately to Biodiversity Offset(s) or Biodiversity Offsetting.

[2] The term Payment for 'Ecosystem' Services is being used mostly as a synonym, but is on aggregate slightly less popular in the literature (Wunder 2015). We use PES to refer indiscriminately to Payment for Environmental Service(s).

logic of PES and BO is thus quite different: the latter is tied to an initial loss of biodiversity; the former to enhanced environmental service provision vis-à-vis a baseline scenario. Ecological equivalence and permanence are typically not key to PES design, yet matter greatly to achieving the stated 'No Net Loss' objective of many BO policies (Maron et al. 2018). As we will argue, this is not just an academic discussion over a miniscule conceptual facet, but has potentially weighty policy implications: misusing the (less strict) PES framework for guiding the (more impacting) BO implementation might ultimately jeopardize biodiversity outcomes. Conversely, there may also be important lessons for BO to be learned from PES.

Thus, our first goal is to clarify the difference between PES and BO from a conceptual perspective, using case examples to illustrate them (Section 2). Indeed, Wunder (2005; 2015) proposed to address the fuzziness of the PES concept by arguing for a fairly narrow definition of PES. Following the same footsteps, we see a need to clarify under what circumstances BO can be considered as PES. Secondly, we attempt to draw lessons from PES to enhance the performance of BO implementation (Section 3). We close with conclusions for practice and further research (Section 4).

2 Defining BO to enable a comparison with PES

2.1 Defining biodiversity offsets

The most commonly used BO definition is from IUCN, cf. ten Kate et al. (2004) that define biodiversity offsets as "conservation actions intended to compensate for the residual, unavoidable harm to biodiversity caused by development projects, so as to ensure no net loss of biodiversity (…) Before developers contemplate offsets, they should have first sought to avoid and minimize harm to biodiversity." (ibid: 13).

Correspondingly, BBOP (2012: 13) states: "Biodiversity offsets are measurable conservation outcomes resulting from actions designed to compensate for significant residual adverse biodiversity impacts arising from project development after appropriate prevention and mitigation measures have been taken. The goal of biodiversity offsets is to achieve no net loss and preferably a net gain of biodiversity on the ground with respect to species composition, habitat structure, ecosystem function and people's use and cultural values associated with biodiversity."

These two definitions are the most frequently used in the academic literature on BO (Bull et al. 2013a; Calvet et al. 2015). These definitions of BO are largely aspirational, in that they hinge on conditions for good (desirable) outcomes. In practice, the desired conditions that BO are to meet greatly differ according to the government regulations or internal (voluntary) policies adopted by corporations and conservation organizations. For instance, the scope of BO can range from biodiversity in general (BBOP 2012) to very specific biodiversity components such as specific protected species, or certain wetland types (Quétier & Lavorel 2011). In the case of voluntary BO, there is seldom any external verification that BO conditions are *de facto* met, beyond the agreed rules between the developer and the BO provider. Desirable socioeconomic outcomes (such as equity among stakeholders and cost-effectiveness of public policies) are often ignored (Bidaud et al. 2017; Griffiths et al. 2018), and ecosystems services generally remain outside the scope of BO regulations (Jacob et al. 2016; Wawrzyczek et al. 2018).

For research purposes, a useful definition should arguably reflect practices across variable ecological and institutional settings. Wunder (2015: 241) underlines the need of isolating desirable outcomes from definitions. For instance, including "additionality" in the definition of forest laws and protected areas would mean that an ineffective forest law or a "paper park"[3] could not be labeled a "law" or a "protected area", respectively. Hence, a PES definition must be consistent and precise enough for generating empirical knowledge, distinctive in function from indirect positive incentives, robust to intertemporal variations in implementation, and simple enough to remember (*ibid*). Some analysts get close to this objective for BO, such as Maron et al. (2012:141-2): "compensating for losses of biodiversity components [or values] at an impact site by generating (or attempting to generate) ecologically equivalent gains, or 'credits', elsewhere (i.e. an offset site)". Following the Wunder (2015) narrow definitional criteria, inspired by Max Weber's "ideal types" (*Idealtypus*), we propose to define BO as:

(1) the supply of an ecological gain

(2) in response to an ecological loss

(3) located in a compensation site distinct from the impacted site

(4) following agreed-upon criteria for the ecological equivalence between gains and losses.

This definition is arguably broad enough to include all types of offsets (both regulatory and voluntary) performed worldwide. It applies to future impacts, assessed *ex-ante* (e.g. in the context of environmental impact assessments and permitting), but also to impacts assessed *ex-post* (e.g. accidents, once they have occurred). It also covers 'averted-loss[4]' and 'restoration' offsets (Bull et al. 2013a) and it excludes scenarios of merely financial compensation. We deliberately do not specify who needs to offset ecological losses, and for which types of events, activities, projects, plans or programs. Nor do we delimit those who can provide ecological gains, and whether they need to hold property or land-use rights. Criterion (3) sets BO apart from other mitigation measures, such as avoidance and minimization (reduction), or the rehabilitation/restoration of the impacted area when operations end (decommissioning). As experience shows, although the underlying principles of BO (1, 2, 3) are simple, how to reach agreements between stakeholders regarding a tentative ecological equivalence (4) can be complex, yet decisive for BO outcomes (Quétier & Lavorel 2011). The meaning of ecological equivalence is hotly debated (Maron et al. 2012), with some authors arguing that like-for-like requirements do not make any sense (e.g. Walker et al. 2009).

2.2 Comparing the BO and PES definitions

Table 1 presents the differences and commonalities between BO, as defined above, and PES, as defined by Wunder (2015): "voluntary transactions between service users and service providers that are conditional on agreed rules of natural resource management for generating offsite services".

[3] A "paper park" refers to an administratively declared protected area that *de facto* enjoys no or only negligible protection: it only exists "on paper" (e.g. Di Minin & Toivonen 2015).

[4] 'Averted losses' offsets rely on ecological gains from the avoidance of impending damages, e.g. through protection measures, rather than from purposely restoring a degraded ecosystem (that would be 'restoration' offsets).

Table 1 – BO vs. PES comparison through their definitions

I. When do BO respect PES criteria?	
Definition of PES	*Applicability of PES criterion to BO*
Voluntary transactions	*For the developer* **No,** if requirements for BO are mandatory **Yes,** if requirements for BO are voluntary *For the environmental service provider* **Yes**, if there is a provider
[Environmental] service user	**Yes,** when the offset is carried out by an external provider **No,** when the offset is carried out by the developer
[Environmental] service provider	
Conditionality based on agreed rules	**Yes** (ecological equivalence)
Generate offsite services (offsite as per PES definition)	**Yes,** if the offset is also carried out offsite (as per the BO definition) – unless the developer has land rights over both sites **No,** if the offset is carried out onsite (as per the BO definition)
II. When do PES respect BO criteria?	
Definition of BO	*Applicability of BO criterion to PES*
Ecological gain	**Yes,** if an environmental service is provided to generate the ecological gain
Ecological loss	**Not Applicable**: usually no ecological loss in PES
Located in a compensation site distinct from the impacted site	**Not Applicable**
Agreed criteria on ecological equivalence between gains and losses	**Not Applicable**

When do biodiversity offsets respect the payment for environmental services definition?

The first part of Table 1 shows that the only PES criterion that BO always complies with is, as in any contract, *conditionality upon agreed rules*: BO is framed in terms of achieving equivalence with a biodiversity loss. A developer may meet BO goals through contracts with several service providers, on several pieces of land: in that case, the conditionality is to reach a given share of the ecological equivalence objective. The other PES criteria are only verified in some instances of BO, as described below.

Firstly, if the BO is implemented by the developer itself, the dichotomy between environmental *service user and provider* becomes obsolete: there is no transaction between actors. But the different steps to implement BO, e.g. requiring the finding, securing and managing of land for conservation are rarely conducted by developers themselves. Typically, they are contracted out to specialized firms, conservation organizations, landowners or users such as farmers. In this situation, the developer is the environmental service user (buyer and beneficiary of the ES) and there is a third-party provider (and seller) of the environmental service that delivers the BO requirements to the developer.

Secondly, in Wunder's 2015 definition of PES, both the service user and provider must *voluntarily* engage in a contract. His paper discusses the collective organization of environmental services providers and/or users and the degree to which they reflect voluntary engagement by individual agents vs. collective action where not all individuals may pre-agree. This is also an issue in the context of BO. Some BO commitments by developers are voluntary in that they are not required by government regulation, but emerge from interactions with stakeholders (Rainey et al. 2015). When government regulations impose BO, these could be considered mandatory. BO providers are most often free to engage or not in a contract, making BO voluntary for the provider, but it is more difficult to describe the voluntary nature of the involvement of the providers in cases where their property or use rights are not well defined. If there are several BO providers, BO users can be said to voluntarily choose their service provider. However, this would not be enough to consider that a mandatory BO is voluntary from the BO user's side, meaning that the transaction is not fully voluntary, as required by the PES definition.

Thirdly, the PES definition mentions that "[environmental] services are generated offsite", meaning that the "ES [environmental services] beneficiaries are thus external to the physical site where ES provision is generated" (Wunder 2015: 242). In other words, the beneficiary does not own or hold property rights to the site where the environmental services are generated, as suggested by Karsenty & Ezzine de Blas (2016). In the context of BO, "offsite" usually means the offset is implemented on another site than the development project footprint[5], without considering land rights: what matters are the ecological criteria for equivalence, permanence, etc. Consequently, only BO carried out on land that the developer does not own or has no property or use rights to can be considered as "offsite", analogous to the PES definition of Wunder (2015). When land is bought by a developer for BO purposes and property rights are transferred to an environmental NGO for e.g. restoration and management activities, the situation still complies with the offsite (PES) definition, provided the developer continues to fund the management of the land now owned by the NGO, and thus keeps "buying" the environmental service. The points made above require a good understanding of property and land-use rights that vary considerably between countries[6]. Where these are not well-defined or conflictive, determining whether a PES is being used to implement BO requirements can be hard to decipher. Figure 1 graphically represents the situations where BO initiatives fit under the PES definition.

[5] However, in the particular example of mining, BO implemented in areas within a mining concession but outside the footprint of the mine could be said to be "offsite". Concessions can be very large, and mining companies generally do not have full land-use rights on the full extent of the concession.

[6] For instance, the common law considers the property rights as a bundle of rights and the roman law considers property rights as a full ownership.

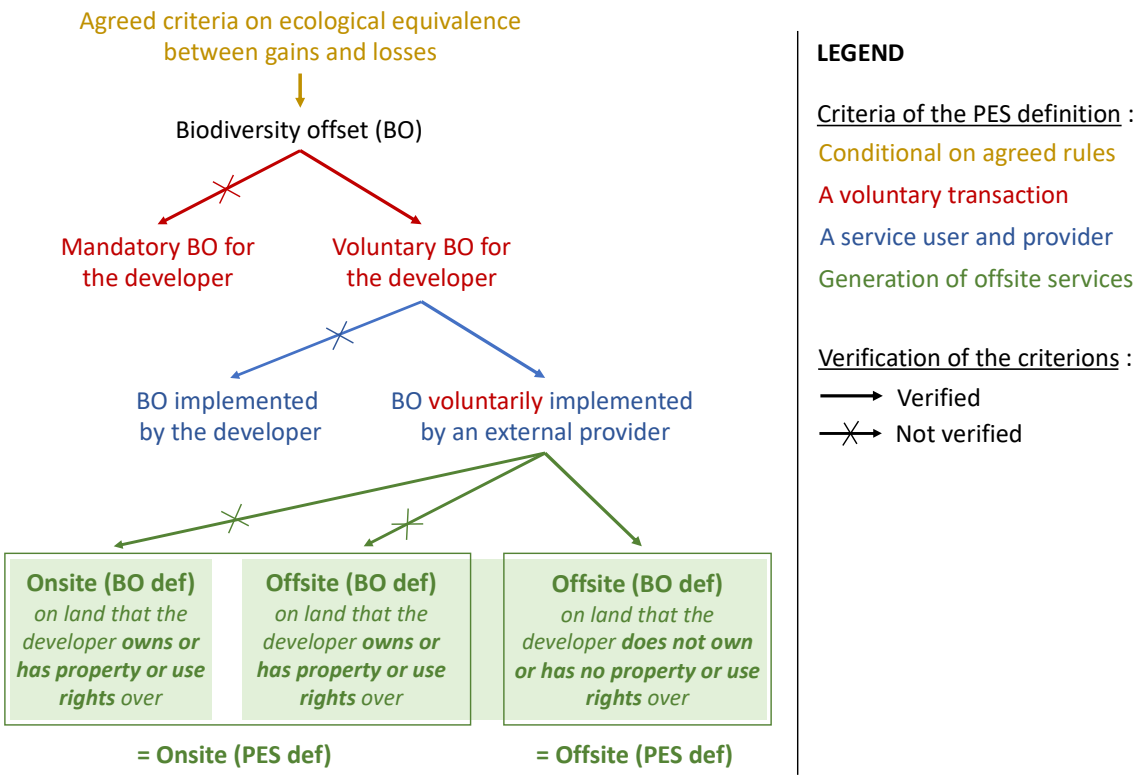

Figure 1 – Verification of PES criterions for the possible BO schemes

When do payment for environmental services fit under a biodiversity offset definition?

The second part of Table 1 shows that PES do not have to comply with three of the four essential features of BO (addressing an *ecological loss*, respecting an *equivalence between gains and losses,* and being *located in a distinct site from the impacted one*) because, unlike BO, the goal of PES is not to address an *ecological loss*. Most of the BO definition is thus purposeless for PES. In PES, the environmental service can involve an ecological gain, but only those related to biodiversity might have a link with BO.

Beyond this comparison of definitions, BO and PES might be said to have different economic founding principles (see Box 1).

Box 1: The economic founding principles of BO and PES

Are BO as utilitarian as PES?
In economics, PES are based on a welfare approach (Muradian et al. 2010) whereby a consumer pays a provider (e.g. a farmer) to deliver a certain level of anthropogenic value (e.g. positive environmental externalities) from ecosystem management (e.g. farming practices). This is clearly utilitarian, with utility (a level of value or welfare) defined as the satisfaction provided by a good or a service to a consumer. On the contrary, BO is focused on providing in-kind ecological gains that address losses to species or habitats, independently of their anthropocentric values, and is thus not based on the utility concept. Our proposed definition, however, allows for utilitarian equivalence criteria to be considered, including the continued provision of certain ecosystem services (Jacob et al. 2016).

Does BO really conform to the polluter-pays principle?
PES are generally tied to the 'beneficiary-pays' or 'provider-gets' principle (beneficiaries reward the maintenance or improvement of an environmental service by someone else). In turn, BO is often tied to the 'polluter-pays' principle whose foundations are in welfare economics (from Pigou and Kaldor-Hicks principle of compensation) and where the objective is to internalize externalities from pollutant activities using taxes, rather than ecological restoration, in order to get a monetary compensation corresponding to the level of loss of values induced by the pollution (Vaissière et al. 2017a). However, not only is BO not necessarily utilitarian in focus, but it excludes strictly financial payments to internalize negative externalities. It would be more appropriate to say that BO follows a 'polluter-restores' principle. Indeed, jurisdictions that allow developers to pay into funds that aim to pool BO resources often have a poor track record (e.g. Narain and Maron 2018) and these arrangements are better described as (green) taxes.

2.3 "Grey areas": when PES and BO share a common design

Using the PES definition of Wunder and the BO definition we have proposed, we show that BO and PES could be similar, in structure, only within very specific contexts and conditions (Figure 1): when a contract is set between a developer (or a collective organization) voluntarily deciding to implement BO and another agent (or a collective organization) voluntarily supplying a gain of biodiversity that is ecologically equivalent to the loss of biodiversity caused by the developer, due to pressure from stakeholders or the willingness of developers to develop and implement an internal set of values (or greenwash their image). And the gain of biodiversity has to be located in a site distinct from the one impacted by the development, and which the developer does not own or has no property or use rights over.

We have chosen four illustrations of previously studied test cases where the distinction between the structure of PES and BO could easily be blurred (Table 2). These are BO schemes, and we scrutinize whether they also hold PES features. Given our findings above (Table 1), conversely analyzing PES examples that are structurally not based on the principle of BO would be meaningless. Some PES schemes certainly share with BO the idea of "offsetting" environmental damages, such as in carbon-based PES initiatives[7]. Yet, the difference between them is that a PES programme does not pretend to assess in detail a specific, discrete underlying damaging activity 'prior to the cure', as certainly is the case for BO interventions.

In terms of the *voluntary nature* of the transaction, our BO examples show that transactions are seldom fully voluntary on the beneficiary side. Many respond to requirements for legal compliance, or anticipations thereof. The projects that are financially supported by lenders who have committed to the Equator Principles[8], e.g. Ambatovy and Ngoyla mining projects, are required to apply the performance standards (PS) of the International Finance Corporation (IFC), and in particular PS6 on

[7] For instance, the Dutch FACE Foundation has financed reforestation in tropical areas, such as the PROFAFOR programme in Ecuador (Wunder & Albán 2008), to offset emissions from Dutch electricity companies. Of course, the focus here is not on biodiversity outcomes, which may be positive (e.g. in restoring highly degraded lands) or more dubious (e.g. when planting exotic species in high-altitude grasslands), but are only collateral to the main focus on carbon. The point is that such voluntary offsets are seen exclusively as a financing mechanism.
[8] The Equator Principles constitute a risk management framework voluntarily adopted by financial institutions for determining, assessing and managing environmental and social risk in project finance. See: https://equator-principles.com

biodiversity and natural resources. PS6 requires that the bank's clients implement a mitigation hierarchy of avoiding and minimizing impacts and offsetting any residual impacts on natural habitats to achieve no net loss or, in the case of critical habitats, a net gain of biodiversity (IFC 2012; 2018). The Ambatovy mining company also voluntarily accepted to become a BBOP pilot, which demonstrates commitment to BO and the goal of achieving no-net-loss and net-gains of biodiversity, and to communicating targets and progress. In addition, Madagascar's rapidly evolving environmental regulations may soon include stronger mitigation and offsetting requirements, so that future compliance contexts were perhaps anticipated. In the Contournement Nîmes Montpellier railway example (henceforth CNM)[9], the farmers are voluntary involved to provide ecological gains for BO through contracts as agri-environmental schemes funded by the developer.

It is notable that PES in practice are also often being applied in policy mixes, where payments are being conditioned upon cross-compliance with pre-existing (and partially overlapping) command-and-control interventions. This includes the European Union's Common Agricultural Policy whereby farmers are already eligible to agri-environmental payments and the emblematic case of Costa Rica's national PES scheme, which has been operated since 1996 (Pagiola 2008). It is thus also a question of whether we can analytically consider a particular environmental intervention as being independent, or whether we should rather see it as a component of an integrated policy strategy.

It can be hard to know if engagement was fully voluntary on the supply side. Landholders could be forced into giving up land or resource access rights to let developers implement offsets, becoming cases of displacement and resettlement from development projects that come with their own local requirements for 'compensation', as in the Ambatovy example (Bidaud et al. 2015; 2017; 2018).In the Ngoyla example, there were no recognized land-use rights within the government-owned forest conservation concession prior to its attribution, but even within the government itself, the decision to assign the land to BO was not straightforward (Ongolo & Karsenty 2015). Finally, as mentioned earlier, the collective organization of the suppliers or providers of the environmental service makes it difficult to be sure that all the individuals agree (free prior and informed consent) with the engagement in a BO programme.

There are always *services users* and *service providers* in our examples, as these were selected on their contract-based nature (entirely or partially). In the CNM example, the originally proposed BO scheme has to be spread in space and time across multiple short-term (5 year) contracts, for the full duration of the developer's 25 year BO obligations. This carries obvious risks (Calvet et al. 2019). In the Ambatovy example, there is a diversity of local stakeholders involved, with NGOs and government agencies acting as intermediaries. In the Ngoyla example, a public protected area agency acts as an environmental service provider to the developer. This contrasts with public agencies acting as buyers of environmental services, as is often the case in government-mediated PES. Interestingly, while in most PES and BO situations service users seek providers, mitigation bankers here anticipate the needs of service users. Beyond bespoke offsets implemented by a developer (with or without a service

[9] The CNM project is a large-scale BO program involving hundreds of farmers in southern France to offset the ecological impacts of a new 80 km high-speed railway line managed by Oc'Via, the developer. For further information see Calvet et al. (2019).

provider), mitigation banking has emerged since the late 1980s to overcome some of the challenges of offsetting impacts on wetlands in the USA.

Regarding *conditionality*, all the contracts in our examples include agreed criteria that are based on the ecological equivalence between biodiversity losses and gains. Of course, these criteria are more or less strict, and not always correctly applied. In the examples of CNM and mitigation banking, loss-gain approaches on ecological functions and/or habitat quality were used to determine the type and size of the BO. Because they are applied within an enforceable regulatory framework, the justification of the selected BO activities, monitoring and enforcement should be relatively high. The other two examples are voluntary BO with weaker governance structures.

Regarding the *offsite* nature of the BO (PES definition), it is easy to determine if the developer (user of the environmental service) has property rights or not on the site where the BO will be carried out. Each of our four examples has at least a part of its BO programme realized offsite. We acknowledge that the uncertainty about the land status on the provider side is often problematic, but the fact that the service provider owns or has property rights over the BO land, or not, is not actually important in defining the offsite nature (PES definition) of the scheme. Rather, we believe that this uncertainty raises more concerns as to the voluntary involvement of providers, as discussed in detail above.

Although not exhaustive, our selected examples illustrate quite contrasting BO situations: it is hard to find actual BO cases where the direct and voluntary requirements of Wunder's PES definition are met. In other words, in spite of the clear family ties, BO and PES are in this respect quite distinct[10].

[10] Of course, alternatively, a broader definition of PES (e.g. Muradian et al. 2010, Froger et al 2016) may well include cases where the voluntary nature of the transaction, on the provider or the user side, is blurred, enabling some BO to be categorized as PES with specific requirements in terms of offsite services. However, the more restrictive definitions of both BO and PES that we have used here are more helpful in differentiating the two instruments.

Table 2 – Features of selected BO examples

	Ambatovy (1)	Ngoyla (2)	CNM (3)	Mitigation banking (4)
Details of the example				
Country	Madagascar	Cameroon and Congo	France	Florida, United-States
Project	Nickel and cobalt mine: extraction site, 220 km pipeline, processing plant on the coast, extension of the port at Toamasina; in operation since 2012	Iron ore mines of Mbalam and Nabeba: extraction sites, infrastructure to transport the ore to the coast 500 km away; the projects were put on hold in 2014 but are now being reconsidered	Railway bypass of 80 km in Southern France called the "Contournement Nîmes-Montpellier" (CNM); construction completed in 2017	Several development projects (infrastructures, housing, beach nourishment, etc.)
Residual biodiversity impact after avoidance and reduction measures	The footprint includes the loss of about 2000 ha of native forest, with risks of additional induced impacts from population influx	Total footprint is around 15,000 ha, including over 2000 hectares of old-growth rainforest, with considerable risks of induced impacts from population influx	About 2000 ha of mainly agricultural land harboring endangered species	Cumulative losses of wetland area and functions
BO program	Averted loss offset: conservation of existing forests in 5 sites totaling more than 18,000 ha. This includes working with forest authorities and a conservation NGO to ensure: - Strict protection from deforestation by ranger patrols - Training of local communities to new rules for local multifunctional use (e.g. establish woodlots within the forests, where wood collection was permitted and managed) - Rural development program to generate alternative revenue streams to compensate for restrictions on forest uses (e.g. training farmers to intensify rice production, introduce poultry farming, etc.)	Averted loss offset: conservation of 164,000 ha of mostly previously unlogged old-growth forest in southern Cameroon. The mining company bid for and won this forest concession as a 'conservation concession', against a competing proposal by Wildlife Works to sell carbon credits in the context of REDD+. It was designated as a Faunal Reserve in 2014 (Ministerial Decree 2014/2383), in agreement with the government, and management handed over to the Ministry of Forests	Restoration offsets: changes in agricultural practices or farmland use to provide favorable habitats for the little bustards. As of April 2019, about 1200 ha of agri-environmental biodiversity offset contracts with more than 100 willing farmers covering over 500 farm plots. In addition, the developer also purchased 512 ha of farmland for conversion into favorable land-cover that was leased to farmers for management.	Mostly ecological restoration but also several averted loss programs: a specialized offset provider, the mitigation "banker", voluntarily undertakes ecological restoration and/or conservation actions on his or her land, to then sell publicly authorized 'credits' to one or typically several developers. When the mitigation bank is sold out (i.e. all the credits have been sold), the land is often transferred to a NGO or another environmental stakeholder to carry out the long term management plan

	The offset program is not yet fully operational			
Lifespan of the BO program	Until 2040	Not specified	25 years through renewable five year duration contracts	Perpetual (conservation easements)
Payment	Funding is organized by the mining company, which (in 2019) is also considered using a conservation trust fund as a vehicle for securing its BO financing	The mining company would cover the cost of managing the Faunal Reserve until the end of its operations. The collapse in the price of iron ore in 2013 put these commitments on hold, and local conservation organizations (e.g. WWF) are now trying to raise funds for the Reserve, including through carbon credits	The annual budget for farmer contractual payments is about 1564 M€, individually ranging from 100 to 548 €/ha/yr. These rates were based on perceived farmer opportunity and direct implementing costs	In Florida, credits range from USD 25,000 to USD 200,000 (average size: 1.3 ha), covering investment costs and profit margin, but otherwise being determined by supply and demand
Criteria of PES				
Voluntary	Users: yes, but the lenders have committed to the Equator Principles and Ambatovy is a BBOP Pilot Project so BBOP's Standard on Biodiversity Offsets must be followed Providers: uncertain due to unclear property and use rights, and strong power asymmetries	Users: yes, but the lenders have committed to the Equator Principles Providers: yes, but the government's position remains unclear and there are uncertain use rights by indigenous peoples	Users: no, required by French law, but freely selected among the farmers Providers: yes	Users: no, required by the US law, but developers can generally choose from which seller to purchase their legally required credits Providers: yes, several scenarios exist, the initiative for a mitigation bank may come from the proper landowner, from a land investor, or from someone signing a joint venture with the landowner to trade credits
Service user	Ambatovy mining project	Mbalam and Nabeba mining projects	Oc'Via, a special purpose vehicle to finance, build and operate the railway line	Various developers
Service provider	Government authorities and national and international NGO partners, acting as intermediaries	The Ministry of forests (public protected area agency), possibly	Farmers and a third party which arranges the transaction between farmers and the developer; this is an *ad*	Mitigation bankers The legal service provider is the sponsor of the mitigation bank

	with local communities, for BO activities outside the concession. BO activities on the concession are carried out by Ambatovy itself	with technical support from an international NGO	*hoc* consortium composed of a regional nature conservancy and land trust group (Conservatoire Régional des Espaces Naturels), a local bird conservation group (Centre Ornithologique du Gard), and the local farmer union-run body providing technical and administrative support to farmers (Chambre d'Agriculture du Gard). This consortium is in charge of designing the BO, selecting the farmers and the plots enrolled and monitoring them for Oc'Via	who can be an individual or a legal entity
Conditionality	Dedicated biodiversity metrics were developed for the project and are used to review progress against no net loss and net gain commitments included in agreements with lenders	No net loss of biodiversity Designation of the concession as a Faunal Reserve that must be managed by the Ministry of forests by decree	Maintain the equivalent of the ecological losses assessed (3279 CU) for 25 years. Rules were used by the consortium to select the plots (e.g. location, expected ecological gain, etc.). Compensation Units (CU) gains depend on the initial cover and more significant changes of land-use generate more CU per unit area: for instance, 2 CU/ha for converting cereals and intensive "improved" grassland to Little Bustard habitat and 2.5 CU/ha for converting arboriculture and vineyards. Farmers are committed to the changes they choose among a catalogue of 11 measures.	No net loss of wetland Specific regulatory framework (mitigation banking instrument, conservation easement, unit sale certificate, etc.): the number and type of credits that mitigation bankers can sell and developers must buy are ruled by regulators. Loss-gain approach based on wetland functions and quality to assess the need, type and size of BO
Offsite services	Yes: several 'off-site' measures (the Ankerana Forest and the forests surrounding the Torotorofotsy Wetlands RAMSAR site) *No: there are also 'on-site' and adjacent activities (forests in the mining concession, its buffer zones*	Yes: the faunal reserve is out of the mining concession	Yes: the farmers' plots are outside of the footprint of the railway	Yes: credits sold to developers that have impacts outside the footprint of the mitigation bank *No: credits used by mitigation bankers for their potential own projects*

	and a part of the Analamay-Mantadia Forest Corridor)			

Sources: (1) Ambatovy 2016, BBOP 2012, Bidaud et al. 2015 and 2017, von Hase et al. 2014, (2) Quétier et al. 2015a, 2015b, Ongolo & Karsenty 2015, (3) Quétier et al. 2015c, Calvet et al. 2019 (4) Levrel et al. 2017, Vaissière and Levrel 2015, Vaissière et al. 2017b

3 Lessons from PES for enhancing BO performance

Despite the differences in structure presented in the previous sections, BO and PES share many common issues and challenges in design and implementation. A substantial literature discusses practical conditions and good practices for implementing BO (e.g. Quétier & Lavorel 2011; BBOP 2012; Bull et al. 2013a; Gardner et al. 2013; Gordon et al. 2015; Moilanen & Kotiaho 2018) and PES (e.g. Ezzine de Blas et al. 2016; Wunder et al. 2008; 2018). They share various preconditions and criteria for good outcomes, such as having relevant scope and spatial targeting (e.g. focusing on areas in urgent need of restoration or facing immediate threats), having measurable performance indicators, enforcing conditionality[11] (monitoring, sanctions for non-compliance, performance-based payments), or embedding actions in large-scale nature conservation approaches to enhance performance (threshold effects) and share costs (e.g. through shared monitoring and evaluation systems, etc.). A desirable characteristic for both PES and BO is payment differentiation (i.e. the price paid for a given action differs between providers) to improve cost-efficiency (Calvet et al. 2019 for BO, Bamière et al. 2013 for PES) and 'additionality' of ecological gains with respect to a defined 'business as usual' baseline, or reference frame including e.g. historical/ projected rates of biodiversity loss, regulatory obligations, political commitments or previously funded actions (see Box 2).

Box 2: the important criteria of additionality in PES and BO

Karsenty et al. (2017) discuss the economic and legal dimensions of additionality for PES, which are very similar to those of BO, and are difficult to achieve in practice. Many PES projects focus on averting further losses by covering opportunity and/or management costs (Gómez-Baggethun & Ruiz-Pérez 2011; Chan et al. 2017), which means a credible counterfactual scenario must be agreed upon. On the contrary, investments in ecosystem restoration are often a hallmark of BO, especially when it is hard to make a case for 'averted loss' gains (see Quétier et al. 2015d). Indeed, offsetting development impacts through the conservation of existing biodiversity ('averted-loss') carries considerable risks of not halting long-term biodiversity loss, e.g. through cost- and motivation-shifting (Gordon et al. 2015; Maron et al. 2012; 2013). Regarding regulatory additionality, Karsenty et al. (2017) relate cases where payments only served to facilitate the introduction of new regulations or prohibitions, making these more politically acceptable.

There is a growing concern that BO may only rarely meet their stated objectives (e.g. Maron et al. 2012; Curran et al. 2014; Lindenmayer et al. 2017; Bezombes et al. 2019) and that they may fail to be well integrated into their local socio-economic contexts (Jacob et al. 2016; Bidaud et al. 2015; 2017; 2018; Griffiths et al. 2018). Although mentioned in good practice guidance, considerations of equity (i.e. BO winners and losers), stakeholder participation (i.e. inclusiveness in BO design and implementation), transparency in contract formulation (i.e. collectively defining the rules of the game)

[11] There is an increasing focus on the conditionality of BO based on the achievement of the goal of ecological equivalence, while conditionality in PES is more often based on proxies (Laurans et al. 2011; Gibbons et al. 2011; Wunder et al. 2008). For instance, the recent French law on biodiversity adopts a result-oriented obligation (i.e. developers must show and monitor the ecological results they provided through BO), instead of a process-oriented obligation, but it is still unknown how this new legal requirement will be implemented in practice. Similarly, species offsets under the US Endangered Species Act are sometimes managed using demographic indicators (Gamarra & Toombs 2018), although this is not a guarantee of overall success (Sonter et al. 2019).

appear to get less attention in BO design and implementation than in PES (a rare example is Griffiths et al. 2018).

Roussel et al. (2019), Vaissière et al. (2018b) and Calvet et al. (2019) analyse the use of contract-based BO with farmers in France, and showed that poor contract design conveys significant risks. The lack of additionality is a well-documented risk and in the case of CNM: Calvet et al. (2019) revealed that 58% of the enrolled farmers in the BO programme declared that they had not made significant changes to their practices. Also, 78% of the farmers contracted out plots on which they were already conducting the targeted agricultural practices revealing they might be benefiting from an undeserved 'windfall effect'[12]. In such cases, poor farmers' selection process (selecting those with the lower opportunity costs) and weak contract enforcement (especially no sanctions and weak monitoring) were both of major importance. Given these risks, PES may thus provide a variety of practical experiences for improving socio-economic design and outcomes from contract-based BO implementation. Below, we discuss in this light three main PES lessons (good practices but also pitfall to avoid) in relation to our four practical BO examples.

Firstly, PES are considered more sustainable if they are permanent, i.e. the induced change of practice/ behavior continues even after payments stop (Karsenty et al. 2017). For instance, Locatelli et al. (2007) found that the Costa Rican PES program has improved local awareness about ecosystem services, even after payments have ended. This element is of particular importance in the context of BO where the ecological losses that must be offset are often permanent. Considering BO contracts as time-bound investments aimed to lastingly transform the behavior and practices of the enrolled stakeholders could provide the necessary long-term ecological gains even after the BO requirements of developers stop. In the Ambatovy example, the developer's rural capacity-building activities could be considered such a long-term investment. In the CNM, contracts are short-term (5 year renewable 5 times). Calvet et al. (2019) found that one-third of the farmers involved in the BO program do not anticipate changing their practices after the contract ends, but one-third will keep them. This raises concerns regarding the permanence of the ecological gains, although short-term contracts also make the BO program more flexible vis-à-vis unanticipated institutional (agricultural policies) and environmental (climate) changes (Calvet et al. 2019; Bull et al. 2013b).

Secondly, motivations and behavioral factors have been widely considered in PES schemes, featuring roles and responsibilities (e.g. Clot et al. 2017, Brimont & Karsenty 2015 in the Malagasy context) and flexible "menus" to meet performance requirements (e.g. Fleury et al. 2015; Andrello et al. 2018). More recently, this question is being studied in BO contexts too. Calvet et al. (2019) revealed the influence of social norms on farmer enrolment into the CNM program. Le Coent et al. (2017) showed that purely biodiversity-framed BO contracts made farmers expect higher payments, flagging sensitivity towards environmental issues. It is widely recognized that BO implementation benefits from the involvement of specialized organizations, dedicated to conservation and more likely to be self-motivated by successful implementation. Under IFC performance standards, for example, project proponents are required to actively engage in partnerships with conservation NGOs for the implementation of their biodiversity actions, including BO. It was the case for Ambatovy which partnered with Conservation International and would have most likely been so for Ngoyla, for which

[12] The 'windfall effect' describes a situation where someone receive a reward for doing nothing additional.

the Cameroonian authorities now get technical assistance from the World Wildlife Fund. These NGOs act as intermediaries with local communities, for which motivations and behavioral factors remain important success factors. In addition, it is worth mentioning that the NGOs themselves are not immune to influence: using BO as a recurrent source of financing for their conservation programs exposes them to a risk of crowding out other sources of funding (Gordon et al. 2015; Maron & Louis 2018).

Thirdly, PES are widely expected to allocate contracts equitably (e.g. Pascual et al. 2010). For BO, more attention should also be given to the socio-economic consequences of area selection, given that similar tradeoffs between efficiency and equity abound (Bidaud et al. 2015). Bidaud et al. (2017, 2018) found some positive impact on local wellbeing in the case of the Ambatovy BO program, but also that timing and beneficiaries were not always matching those suffering negative impacts. In the mitigation banking example, aggregation of BO supply in a single 'bank' sometimes led to the impacted areas and 'banks' being quite distant (e.g. averaging 40 km in Florida - Levrel et al. 2017). The potential redistribution effect in terms of ecosystems services between users of the impacted site and beneficiaries from the restored one must receive due consideration (Vaissière et al. 2017b).

Agri-environmental PES schemes, land trusts and easements (Merenlender et al. 2004), public-private partnerships for conservation (Thackway & Olsson 1999; Saporiti 2006), community-based conservation and other forms of 'private' protected areas (Carter et al. 2008) can all potentially provide lessons for BO implementation. But BO come with specific requirements that must be remembered when investigating implementation options. Due diligence is needed to ensure that BO is not interpreted as just another financing mechanism (e.g. Maron et al. 2016).

4 Conclusion

Biodiversity Offsetting (BO) and Payments for Environmental Services (PES) are often considered jointly as innovative ways to tackle the ongoing biodiversity crisis, and BO are by many observers considered generically as a type of PES, i.e. falling under the same umbrella of intervention. We have argued above that this would generally be a misuse: compared to the most common definition of PES (Wunder 2015), there are few circumstances in which BO can structurally be considered as PES, especially because the PES inherent voluntariness criterion from both sides is hardly ever respected in BO. In terms of our initial intention of clarifying the family ties, we can thus conclude that PES and BO are neither twins nor siblings; they are probably best compared to two fairly "distant cousins": some founding material is clearly shared between them, while other of their respective characteristics are fundamentally different.

Why then is it fairly common that PES and BO are put under the same umbrella? This perhaps forms part of the confusion around the so-called "market-based instruments" (Pirard & Lapeyre 2014; Gómez-Baggethun & Muradian 2015), an arguably excessively broad term under which many instruments constitute economic incentives or resource allocation mechanisms in which markets properly defined actually play no role (Pirard 2012; Wunder 2013). Hence, the scope for conceptual confusion is noteworthy. By claiming to directly offset residual damages from a directly paired environmentally polluting impact from development activity, BO interventions thus come to share much more in the responsibility of the paired loss impacts than would be the case for PES: a PES intervention is generally deemed to be successful if it manages to produce more environmental

services than a business-as-usual baseline, but it is not directly (though sometimes indirectly) used to justify environmental losses prior to the remedying action. No wonder, then, that we need stricter standards for the aggregate environmental benefits of BO than we do for PES schemes: we need to make sure with much more vigor that on balance the adopted impact compensation mechanism is not tantamount to a greenwashing operation.

On the other hand, the design and implementation of contract-based BO can indeed draw valuable lessons from PES experience. Contract design and enforcement matter for successfully converting developers' commitments into measurable conservation outcomes. Also, PES social impacts, safeguards, and side-objectives have been more holistically looked into than for BO. Hence, as BO implementation is continuously expanding, it would be unwise to turn a blind eye to PES.

Still, care is also needed to avoid invalid extrapolations from blindly transferring lessons from one tool to the other. Neither PES nor BO are 'silver bullets' for biodiversity conservation and important organizational and legal limitations need to be considered when choosing a contract-based BO approach and the relevance of lessons from PES must be assessed for the specific purposes of BO. In this context, practical research on the contracting of BO implementation to third parties is also needed so that responsibilities are not lost or diluted, and monitoring and enforcement are not rendered impossible.

The implications of BO failure for biodiversity are stronger for BO than for PES, given that the former entails a direct, and often locally irreversible frontloaded loss of biodiversity, while the latter can be flexibly be interrupted whenever the parties would no longer agree on a service provision contract. Development projects with impacts on biodiversity can go forward on the basis of expected BO outcomes that can be made insecure when dependent on voluntary contracts with land-owners and land-users, as in a PES scheme. If a PES scheme has not been specifically designed for biodiversity purposes, it also cannot be used as a BO setup. The differences in the founding principles of PES and BO matter and safeguards are needed to avoid the risks of uncompensated biodiversity loss that could arise from confusing BO and PES. This is why it was important to clarify how they differ.

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