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# Tackling limitations in biodiversity offsetting? A comparison of the Peruvian and French approaches

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## ABSTRACT

Offsetting schemes to compensate biodiversity loss resulting from land-use change (e.g., urbanization, infrastructure expansion) suffer limitations, related notably to the requirement for ecological equivalence between losses and gains, which cover ecological, spatial, temporal and uncertainty considerations. Such limitations impair the effectiveness of biodiversity offsets. Biodiversity offsetting is nevertheless adopted by an ever-increasing number of countries. We analyze how Peru and France approach biodiversity offsetting and whether and how they address all or some of these limitations, which could serve to inform other countries adopting such mechanism. We show that, although both countries apply similar principles, their no net loss (NNL) objective differs (NNL of biodiversity and ecosystem functionality in Peru vs NNL of biodiversity in France) with consequences on the ecological equivalence approaches adopted. In Peru, the imposed assessment method is habitat-based and adapted to specific ecosystems. By contrast, there is no mandatory assessment methods in France and, with the exception of wetlands, the focus is strongly on protected species, and on species functional traits rather than ecosystems in their entirety. The Peruvian method does not systematically integrate the landscape context and temporal losses are not accounted for, whereas uncertainty could be considered as indirectly taken into account. In France, landscape connectivity is not necessarily included in assessment methods, although it can be taken into account in practice. Further, although weighting assessment methods may address temporal losses and uncertainty, their variety prevents a comparison of outcomes. Additional elements would warrant further analysis (e.g., monitoring and compliance).

## KEYWORDS

Compensation, biodiversity offsets, ecological equivalence, assessment methods, temporal losses

## 1. Introduction

The primary driver of biodiversity loss is land-use change, which results notably from urbanization and infrastructure expansion (IPBES 2019). Compensation for environmental damage is one way to address such biodiversity loss, and one tool increasingly being used is biodiversity offsets (GIBOP 2019). Biodiversity offsets are measures designed to compensate for significant residual adverse biodiversity impacts arising from project development, as the last step of the mitigation hierarchy (avoid, minimize, restore, and then offset) (BBOP 2009; Bull and Strange 2018). The principle of 'no net loss' (NNL) usually underlies biodiversity offsetting policies (BBOP 2009; Bull and Strange 2018). Offsetting is considered *ex-ante* ecological compensation (authorized before the damage occurs, e.g., as part of the environmental permitting process), as opposed to *ex-post* compensation that applies after occurrence of the damage (Lucas 2015), such as in the case of remediation measures required under the European Environmental Liability Directive (ELD) (European Union 2004).

Offsetting is based on the principle of ecological equivalence between impact and offset areas, which requires calculation of losses and gains (Carreras Gamarra et al. 2018), carried out on the basis of indicators (or 'metrics' or 'currencies') chosen as surrogates for biodiversity (Gibbons and Lindenmayer 2007; Quétier and Lavorel 2011; Bull et al. 2013; Maseyk et al. 2016; Bezombes et al. 2018). Metrics can be composite units of aggregated measures of biodiversity or can be more disaggregated by individually accounting for each measured biodiversity element of interest (Maseyk et al. 2016).

Despite its ever-increasing use, biodiversity offsetting suffers, nonetheless, important limitations. A review of English-language peer-reviewed literature (zu Ermgassen et al. 2019) and examples in countries as diverse as Uzbekistan (Bull et al. 2013), France (Calvet et al. 2015), Australia

(May et al. 2017; Lindenmayer et al. 2017) disputed the efficiency and effectiveness of biodiversity offsets. In particular, calculating the biodiversity gains required to achieve ecological equivalence, and hence NNL, is key to designing and monitoring the effectiveness of biodiversity offsets (Carreras Gamarra et al. 2018). Ecological equivalence is, however, one of the main conceptual challenges identified in offsetting-related scientific literature, together with, among others, metrics and location of offsets (Gonçalves et al. 2015). There are concerns regarding notably the choice of metrics to demonstrate equivalence between biodiversity losses and gains (Bull et al. 2013; Gardner et al. 2013; Calvet et al. 2015). To some authors, NNL is not achievable in practice, as it does not apply comprehensively to ecosystems, but rather to values defined in offsetting metrics (Maron et al. 2012; Gardner et al. 2013; Moreno-Mateos et al. 2015). Consequently, offsets offer only poor or incomplete replacement for the loss of biodiversity (Quétier and Lavorel 2011).

Maron et al. (2012) further identify various factors limiting the technical success of offsets, among which time lags and uncertainty. Time lags correspond to the time that lapses from the occurrence of the impact till the offset is fully effective, whereas uncertainty refers to risk of offset failure. Time lags and uncertainty considerations should be integrated in the design of biodiversity offsets, as immediate loss is certain whereas future gain is uncertain. Immediate loss cannot, therefore, be offset by only hypothetical equal gains (Moilanen et al. 2009; Quétier and Lavorel 2011; Maron et al. 2012; Laitila et al. 2014). One method that could be used to account for these time lags is time discounting (Maron et al. 2012; Laitila et al. 2014).

Equivalence assessment methods thus need to take into account various key considerations (Quétier and Lavorel 2011; Bull et al. 2013), summarized by Bezombes et al. (2017) as ecological (target biodiversity and related indicators), spatial (landscape context), temporal (time lags), and uncertainty (risk of offset failure) considerations. Despite hundreds of methods to assess and calculate biodiversity losses and gains (Lavorel 2020), and hence their equivalence (McKenney and Kiesecker 2010; Quétier and Lavorel 2011; Bull et al. 2013, 2014; Marshall et al. 2020), none is entirely satisfactory (Bezombes et al. 2017; Carreras Gamarra et al. 2018). Accounting methods can strongly influence the biodiversity outcomes of an offsetting strategy (Carreras Gamarra et al. 2018). Maron et al. (2012) mention poor measurability of the value being offset as an important issue.

In addition, offsetting schemes are country-dependent: impacts vary depending on the local context and regulatory requirements play an important role in the selection of metrics (McKenney and Kiesecker 2010; Carreras Gamarra et al. 2018). Bull et al. (2014) demonstrated, on the basis of a case study in Uzbekistan, that different equivalence assessment methods from different countries, despite having a common NNL requirement, result in divergent outcomes for biodiversity, highlighting how variable interpretations of NNL can be.

Although NNL appears to be an unreachable objective from an ecological standpoint, every biological entity being unique and hence, by definition, irreplaceable (Devictor 2018), this objective underlies most biodiversity offsetting policies, with the corollary principle of ecological equivalence between losses and gains. The use of biodiversity offsetting is unlikely to subside, in the light notably of its recent identification as an 'innovative scheme' to increase the level of financial resources available for biodiversity conservation (CBD 2022). As a growing number of countries is adopting this mechanism, it is important to ensure that limitations associated with offsetting are properly addressed in policies adopted and implemented at national level. When devising their offsetting schemes, countries do not start *ab nihilo* but tend to study, analyze and assess what other countries have implemented. In this paper, we compare the approaches adopted in two countries that are different regarding their legal approach of biodiversity offsetting, the availability of ecological information and the characteristics of their natural systems: Peru and France. Comparing two very different contexts, we analyze whether, and if so how, these countries have addressed (or at least attempted to address) limitations identified in the scientific literature in relation to biodiversity offsets, and whether it may contribute to rendering biodiversity offsets more effective. This comparative analysis gathers information useful for other countries that have adopted or are currently adopting biodiversity offsetting as a mechanism. They may also inform the study countries on possible paths for improvement.

## 2. Materials and Methods

Peru and France were selected as study countries on the basis of various elements. First, their legislations on biodiversity offsetting have enough similarities to enable a comparison of their elaboration and implementation. Second, in both countries, many biodiversity offsetting-related developments have occurred since the early 2010s.

### 2.1. General legal framework in study countries

#### 2.2.1. Peru

In 1990, Peru adopted its Environment and Natural Resources Code, which establishes the prevention, remediation and polluter-pays principles (Peru 1990). Peru's Constitution, adopted in 1993, stipulates that 'the State is obliged to promote the conservation of biological diversity' (Peru 1993). In 2005, the country adopted the General Environmental Act, which provides that 'when it is not possible to eliminate the causes that generate [environmental degradation], mitigation, recovery, restoration or eventual offsetting measures are adopted, as appropriate'; offsetting measures are to be implemented only as a last step (Peru 2005). Further, the General Environmental Act provides that, where human activities are likely to cause significant environmental impacts, they are subject to the National Environmental Impact Assessment System (SEIA, *Sistema Nacional de Evaluación de Impacto Ambiental*). The act through which the SEIA was created notably refers to the biodiversity offsetting plan that the environmental impact assessment (EIA) of certain projects must include (Peru 2001). The SEIA did not, however, come into force until 2009 and the adoption of the SEIA Regulations (Peru, MINAM 2009). In 2015, Peru passed a new law, which enacts measures to optimize and strengthen the SEIA (Peru 2015).

Projects subject to environmental licensing (*certificación ambiental*) are classified in three categories, on the basis of the severity of their potential impact on the environment (light, moderate or significant): (i) category I (Environmental Impact Statement), (ii) category II (semi-detailed EIA "EIA-sd") and (iii) category III (detailed EIA "EIA-d") (Peru 2001). Category III projects are 'projects whose characteristics, size and/or location may produce significant negative environmental impacts, quantitatively or qualitatively, requiring an in-depth analysis to examine their impacts and propose the corresponding environmental management strategy' (Peru 2001). Only category III projects are to include a biodiversity offsetting plan, and hence implement offsetting measures, as the case may be (Peru 2001).

The Peruvian Environment Ministry (MINAM, *Ministerio del Ambiente*) adopted specific guidelines and guidance documents in relation to biodiversity offsets: (i) Guidelines for the implementation of biodiversity offsetting under SEIA (Peru, MINAM 2015), (ii) General guide to the biodiversity offsetting plan ('General Guide') (Peru, MINAM 2016a) and (iii) Complementary guide for biodiversity offsetting regarding high Andean ecosystems (*tolar*, *pajonal de puna*, *césped de puna*) (Peru, MINAM 2016b). The focus of the first complementary guide on high Andean ecosystems may be explained by the fact that they correspond to 14% of the national area and cover 70% of the Andean region, and are where a large part of the mining and livestock activity takes place (Peru, MINAM 2016b). In addition, the *pajonal de puna húmeda* is one of the most threatened ecosystems because of land degradation (Peru, MINAM 2019a). Starting in 2019, complementary guides were adopted for other ecosystems: *bofedales* (Andean wetlands), dry forests, and *yunga* (mountain rainforest ecosystems) (Peru, MINAM 2019b, c, d). Additional guidance documents were adopted for the elaboration of the environmental baseline, and for the identification and characterization of environmental impacts (Peru, MINAM 2018a, b, c, 2022a, b). MINAM also adopted a resolution which approves the prepublication of (draft) guidelines for the elaboration of the Environmental Management Strategy (Peru, MINAM 2020). There are nonetheless no published guidelines on how to apply the mitigation hierarchy.

In Peru, biodiversity offsetting is guided by the following principles: (i) compliance with the mitigation hierarchy (avoid, minimize/reduce, rehabilitate, and offset), (ii)>NNL of biodiversity and ecosystem functionality, or even net gain, (iii) additionality, (iv) ecological equivalence (offsetting must

be in kind) and (v) sustainability of the offset (Peru, MINAM 2015). Although the Peruvian biodiversity offsetting guidelines accept the principle of 'trading up' or 'like-for-like or better' (*igual o mejor*), it is not applied in practice as MINAM is, reportedly, yet to elaborate and adopt a standard on the matter.

### 2.2.2 France

In France, the offsetting obligation was first established in 1976 through the adoption of the Nature Protection Act. The 1976 Act introduced EIA for certain projects and required the application of the mitigation hierarchy (France 1976). However, the offsetting obligation now appears in numerous regulations (on EIAs, urban planning documents, Natura 2000, water law, protected species, etc.). It is not until the late 2000s that the offsetting obligation started being implemented in practice (Semal and Guillet 2017). From the 2010s onward, France elaborated numerous guidance documents on the mitigation hierarchy, referred to as ERC (*Éviter-Réduire-Compenser*, standing for Avoid, Minimize/Reduce, and Offset). These documents, although not binding in nature, include the ERC doctrine in 2012 and the first ERC guidelines in 2013 (MEDDE 2012; CGDD 2013). France later clarified the principles that apply to biodiversity offsetting, through the adoption in 2016 of the Biodiversity Act (France 2016). Since then, the mitigation hierarchy is considered a component of the principle of preventive and remedial action, which implies 'avoiding damage to biodiversity and the services it provides; failing that, reducing the extent of such damage; and finally, offsetting damage that could not be avoided or reduced, taking into account the species, natural habitats and ecological functions affected'. Initially, the first drafts of the bill referred to 'ecosystem services and functions affected' (Alidor 2017). The deletion was, reportedly, to avoid confusion between functions and the notion of ecosystem services (Fèvre 2017).

Biodiversity offsetting applies in the framework of a project's licensing procedure (*ex-ante* ecological compensation). Some form of biodiversity offsetting may, however, also be required under other regulations for accidental damage. Such *ex-post* ecological compensation is required under the national legislation transposing the ELD. Further, the 2016 Biodiversity Act introduced ecological damage ('*préjudice écologique*', initially established by the *Erika* case law (Cour de cassation 2012)) into the French Civil Code, which defines it as 'a non-negligible damage to the elements or the functions of the ecosystems or to the collective benefits drawn by man from the environment' (article 1247), for which compensation 'is primarily provided in kind' (article 1249). There is no imposed method in France to assess ecological equivalence (Bezombes et al. 2018). Following the adoption of the Biodiversity Act, additional non-binding guidance documents were elaborated. Some are of specific relevance to biodiversity offsets, such as the (i) National method for assessing wetland functions (Gayet et al. 2016a, b), (ii) Guide on methods for designing offsets for watercourses (CGDD et al. 2018), (iii) Guide to help define ERC measures within the environmental assessment process (CGDD 2018a), and (iv) Guide for a standardized approach to the dimensioning of biodiversity offsets (Andreadakis et al. 2021).

Under French law, the principles that *ex-ante* biodiversity offsetting must abide by are: (i) objective of NNL, or even a net gain, of biodiversity, (ii) ecological equivalence, and (iii) proximity to the impacted site (geographical and functional proximity). The principle of additionality is not expressly mentioned, but is considered implicit as a consequence of the NNL objective (Dupont and Lucas 2017). Biodiversity offsets measures are subject to an obligation of results and must 'be effective for the entire duration of the damage'. If the mitigation hierarchy cannot be applied satisfactorily, the project should not be authorized as it stands.

### 2.2. Methods

We use the key considerations listed by Bezombes et al. (2017) as criteria against which to evaluate each national policy, and in particular their equivalence assessment methods. We thus based our analysis on four main criteria: (i) ecological (targeted biodiversity and metrics used to assess equivalence), (ii) spatial (location of offsetting sites and landscape context), (iii) temporal (time lags) and (iv) uncertainty (how the risk of failure of the offset is accounted for).

We based the collection of data on a mixed method. For each country, as a first step we conducted a thorough literature review. The reviewed material included primary legal sources (legal texts), secondary sources, as well as grey and academic literature in conservation science and EIA. For France we added a desk review of a local road widening project. The second step included, for both countries, the conduct of expert interviews (Van Audenhove and Donders 2019). The interviewees were chosen among the main nationwide stakeholders in Peru; in France, they were selected among the stakeholders of the road widening case study. The interview guidelines for expert differ from one country to another in the light of the mandatory nature of the equivalence assessment method in Peru vs its non-mandatory nature in France, where a different method can be defined for each single project.

From December 2021 to February 2022, we interviewed 12 individuals with biodiversity offsetting-relevant expertise in Peru. Interviewees were identified on the basis of the literature review and/or recommendations. The list of questions was sent prior to each interview; the questions aimed to gather information on the applicable legislation and its implementation requirements (see Online Resource 1). The interviewees belonged to the following categories: non-governmental organization, consultancy, academia, government. We were unable to interview representatives of the industry association and environmental authority categories, as they did not reply positively to our requests.

The French local road widening project case study was conducted between June 2021 and May 2022. This project is the conversion of an 88-kilometer stretch of the existing *Route Centre Europe Atlantique* (RCEA) into a highway (A79) in the Allier department. We analyzed the biodiversity offsetting process in the environmental licensing procedure, which was later complemented with 11 interviews, carried out from July to November 2022. The interviewees were individuals involved in the RCEA project; they pertained to the following categories: concession company, consultancy, State services, operator in charge of implementing biodiversity offsets. It enabled us to collect information on the elaboration of the offsetting measures in relation to the mitigation hierarchy, and in particular the method proposed to ensure biodiversity gains are equivalent to losses.

In order to comply with personal data protection regulations, the names of interviewees and the entities to which they pertain are kept confidential.

With this method we were able to gather input on the equivalence assessment methods and whether limitations usually associated with biodiversity offsetting were appropriately tackled. We evaluated and compared the objectives and principles relating to biodiversity offsets adopted in Peru and France, as well as the accepted methods to assess and evaluate biodiversity losses and gains to ensure their equivalence, in the light of the four key considerations previously mentioned (ecological, spatial, temporal and un certainty).

### **3. Results**

#### *3.1. Ecological considerations*

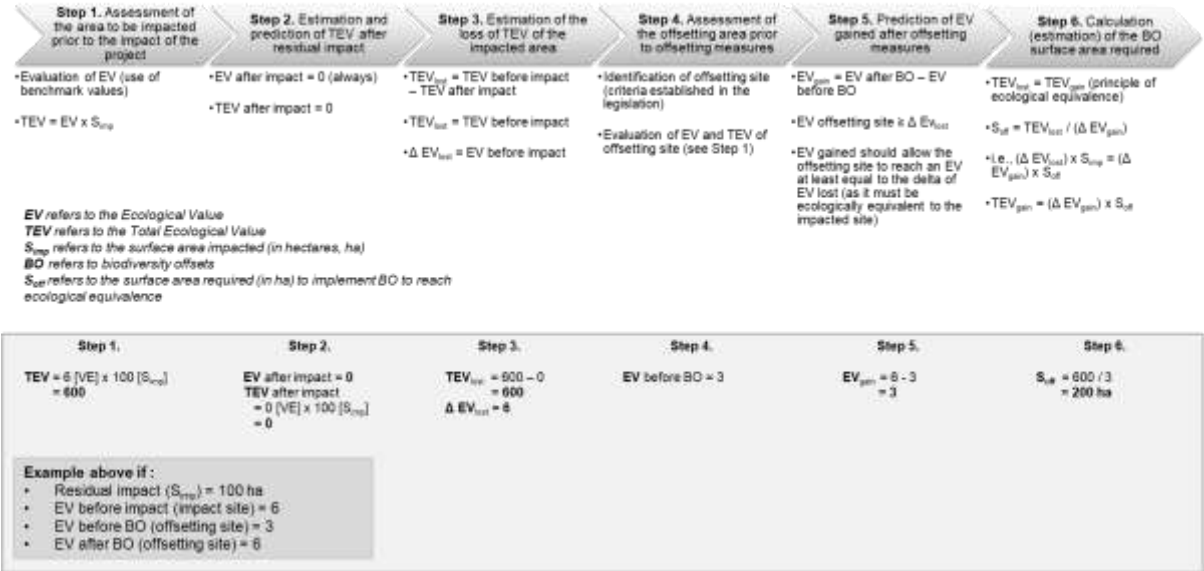
**Peru.** An ecosystem approach is required under Peruvian law. The EIA must be based on a project's level of impact 'on the capacity of ecosystems to maintain their biodiversity and functionality' (Peru, MINAM 2016c). The gathering of information for the baseline must hence 'be oriented towards the identification and characterization of key aspects for the functioning of the ecosystem', with a necessity to identify the trajectory with and without the project (Peru, MINAM 2016c). This approach is further emphasized with the objective of NNL of biodiversity and ecosystem functionality (Peru, MINAM 2015). Ecosystem functionality is defined as 'the dynamic and interdependent process between ecological communities, their space and people, in which their different components, cycles and flows of matter, energy and information are linked, in a landscape context, to ensure ecosystem integrity. This process includes the ecosystem's stability and capacity of evolution, as well as its capacity to generate ecosystem services' (Peru, MINAM 2015).

The application of the principle of ecological equivalence entails that a baseline of the impacted area, as well as of the offsetting area(s), be carried out in order to assess losses and gains. In Peru, the complementary guides on offsetting specify the applicable method for the ecosystems they cover. For

those ecosystems not yet the object of a guide, 'other valid methods may be applied' in compliance with the criteria set out in the General Guide and the biodiversity offsetting guidelines (Peru, MINAM 2016a). In that regard, the complementary guide for high Andean ecosystems provides that the methodology it establishes can be used as a reference for other terrestrial ecosystems (Peru, MINAM 2016b). Interviews confirmed that this guide was indeed taken as a reference and adaptations were made for aspects that it did not consider.

The Peruvian method for estimating losses and gains is based on the Total Ecological Value (TEV), which depends on the calculation of the Ecological Value (EV). The EV is defined as 'the value and/or weighting that represents the conservation status of an ecosystem', and is based on the identification and measurement of ecosystem components and functions (Peru, MINAM 2016a). The complementary guide for high Andean ecosystems establishes a process for calculating losses and gains (Fig. 1).

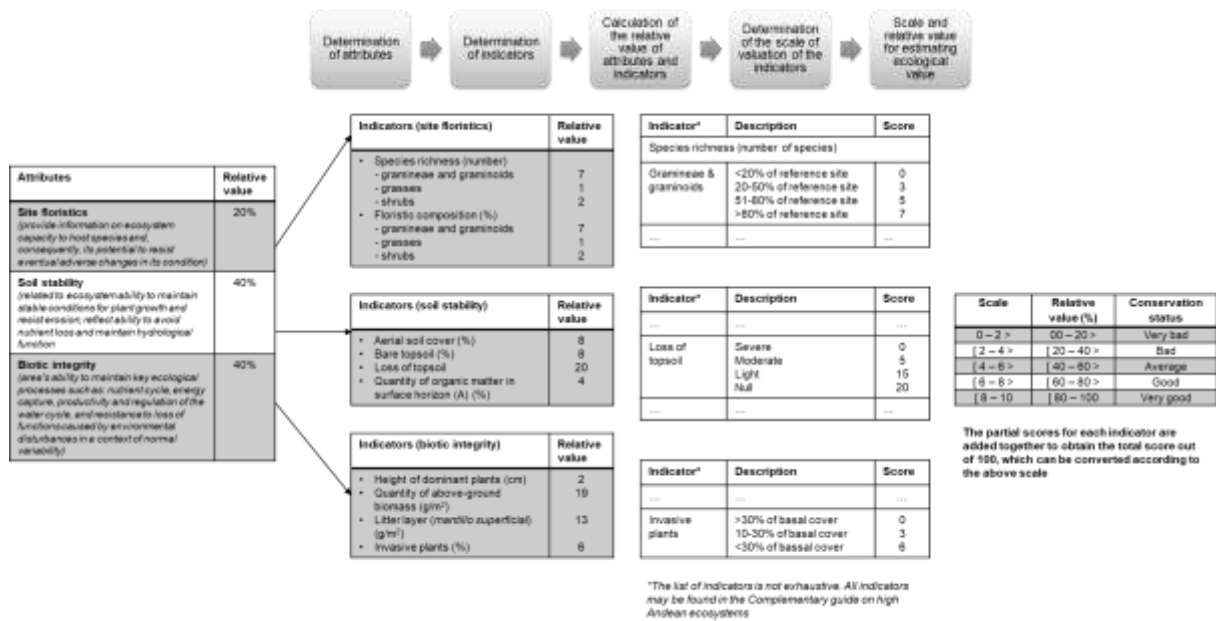
**Fig. 1 Process for estimating biodiversity losses and gains for high Andean ecosystems (based on information contained in Peru, MINAM (2016b, a))**



The conservation status (EV) of the ecosystem is based on a set of attributes and indicators, 'which express the ability of the ecosystem to perform key ecological functions, such as: energy flow, nutrient cycle, hydrological cycle, as well as its ability to recover from alterations caused by disturbance factors' (Peru, MINAM 2016b). The methodology for calculating the EV for high Andean ecosystems, as described in Fig. 2, is built on a qualification system based on three fundamental ecosystem attributes: (i) site floristics, (ii) soil stability and (iii) biotic integrity.

**Fig. 2 Estimating the ecological value (EV) for high Andean ecosystems (based on information contained in Peru, MINAM (2016b)). Multi-criteria matrices, which are based on a hierarchical analysis, were used to calculate the relative value of attributes and indicators. The relative value of each attribute and indicator is the maximum value of the ecosystem**

conservation status scale relative to the benchmark site (benchmark sites having the highest scores). The indicators' rating scale was then determined, based on the relative values assigned to each



The EV calculation requires having ecological reference values obtained from sites considered to be in the best existing conservation status (benchmark site) (Peru, MINAM 2016b). The complementary guide for high Andean ecosystems includes reference values for benchmark sites (Peru, MINAM 2016b). To elaborate these reference values, it was reported that MINAM worked with experts from the *Universidad agraria nacional La Molina*, who had national records of the evolution of these ecosystems over several decades. Field investigations were conducted to validate and refine the general database. The reference points in the guidance document are national averages. The complementary guide does not, however, distinguish between wet *pajonal* and dry *pajonal*. It was stated by an interviewee that, in order to make such a distinction, further research would be required.

Furthermore, although the EIA baseline must include information on fauna for all taxonomic groups, and management plans must be elaborated for impacted species of conservation concern (Peru, MINAM 2018a), the attributes and indicators established in biodiversity offsets guidance documents do not refer to fauna. One interviewee reported that, during discussions on the elaboration of guidance documents, indicators on fauna were discussed; a consensus could not, however, be reached. The work on such indicators is, hence, still pending. One of the reasons advanced by several interviewees is the lack of sufficient (historical, abundant) information to establish reference data to estimate losses and gains. One interviewee indicated that the view generally adopted was that by reaching flora and soil stability, the fauna would also reappear. It was also indicated that the petitioner may propose additional attributes/indicators to calculate losses and gains, which could then relate to fauna. Petitioners are reportedly not keen on including fauna attributes in the biodiversity offsets formula, as it would bound them to measure the value when there is no certainty of the species returning.

Finally, apart from the complementary guides, some interviewees reported a lack of guidance regarding how the offsetting plan is to be implemented. As an example, the guide on high Andean ecosystems provides that offsetting measures must be determined individually, in accordance with the specific situation. The document adds that 'typical measures or strategies to improve the ecological value of high Andean ecosystems include pasture and water management' (Peru, MINAM 2016b), without further indications.

**France.** In France, as previously stated, the objective is one of>NNL of biodiversity, with a requirement of ecological equivalence. There is, however, no mandatory equivalence assessment method. A number of documents have nevertheless been elaborated to guide the sizing and



implementation of biodiversity offsets. The Guide for a standardized approach to the dimensioning of biodiversity offsets classifies the methods into three main categories: minimum ratio (application of a predefined multiplier to a metric – e.g., surface or linear area); weighting (separate quantification of losses and gains by applying ‘loss’ and ‘gain’ coefficients); and ecological variations (*méthodes d’équivalence d’écart de milieux* : separate quantification of biodiversity losses and gains, but using the same metrics, and verification of their equivalence) (Andreadakis et al. 2021). The assessment method may be chosen freely, provided the same method is used to assess losses and gains, and the chosen method is explained and justified (Andreadakis et al. 2021). There is thus no uniform method: it will vary depending on the consultancy (each firm tend to develop its own method) and the specificities of the case (Levrel 2020).

In the case study we carried out for France (RCEA), the methods used were a combination of minimum ratios and weighted equivalence assessment. For wetlands, the petitioner applied the predetermined ratio of 1:1 or 1:2 established in the applicable Water Development and Management Master Plan (SDAGE, *Schéma directeur d’aménagement et de gestion des eaux*) (Agence de l’eau Loire-Bretagne and DREAL de bassin Loire-Bretagne 2015). Under the SDAGE, ‘offsetting is primarily aimed at restoring wetland functionalities’, through the recreation or restoration of wetlands that are cumulatively (i) functionally equivalent, (ii) equivalent in terms of biodiversity quality, and (iii) within the watershed of the water body. If those three cumulative conditions are met, the applicable ratio is (minimum) 1:1; if not, the ratio is 1:2. The assessment of wetland functionality was implemented according to the national wetland functions assessment methodology (Gayet et al. 2016a, b). This method identifies three main interdependent functions (hydrological functions, biogeochemical functions and functions related to species life cycle), whose importance is determined by the calculation of indicators. These indicators include information on soil (e.g., soil acidity, organic matter in topsoil). In addition, a weighting equivalence assessment method was used for non-wetland offsetting, with a focus on habitats and target species. The reference in the legislation to ‘ecological functions affected’ appears to be understood, in the environmental licensing application, as a reference to target species functional traits (i.e., ecological corridor, resting, feeding and reproduction areas), with the exception of wetlands, where an assessment of functional loss is required, as stated above.

### 3.2. *Spatial considerations*

**Peru.** Under Peruvian law, biodiversity offsets must be implemented in an area ecologically equivalent to the impacted area (Peru, MINAM 2015). Offsetting areas are thus required to be ‘natural ecosystems that maintain biodiversity and a potential for ecological values or attributes similar to those of the areas that were impacted by the project’ (Peru, MINAM 2015, 2016a). It does not mean, however, that the impacted and offsetting areas must have the same ecological value before the offsetting plan is implemented (Peru, MINAM 2016a). The offsetting site must be located, as a priority, within the affected ecosystem. If that is not possible, it must be located within the project’s area of direct influence and, as a last resort, in the area closest to the project’s area of direct influence (Peru, MINAM 2016a).

The landscape context is mentioned in the definition of ecosystem functionality (Peru, MINAM 2015). In addition, the biodiversity offsetting guidelines indicate that ‘the areas where biodiversity offsets are implemented must consider the context of the landscape and the range of variation of its elements to ensure its ecological viability and sustainability’ and also refers to ecological connectivity as an element to take into account when choosing offsetting sites (Peru, MINAM 2015). However, landscape-related attributes and indicators are not necessarily included in all complementary guides on biodiversity offsetting. As an example, the complementary guide on high Andean ecosystems does not mention the landscape context, when the guide applicable to *bofedales* list landscape alterations as an attribute (together with water condition, soil condition and biota condition) (Peru, MINAM 2016b, 2019b).

**France.** In France, biodiversity offsets must be ‘implemented as a priority on the damaged site or, in any case, in its vicinity in order to guarantee its functionalities in a sustainable manner. The same measure can offset different functionalities’ (Environmental Code, article L.163-1 II). A new paragraph was added following the adoption of the 2021 Climate and Resilience Act (France 2021), whereby

biodiversity offsets must be implemented as a priority within the preferential renaturation areas identified in land-planning documents. These new provisions entered into force in December 2022 (France 2022), but renaturation areas will not be identified until the relevant land-planning documents are revised.

In the RCEA case study, the calculation of gains in the equivalence assessment method includes a proximity coefficient, which is based on the distance between the offsetting and impact sites, 'in relation to the ability of the target species to move to and colonize the offsetting sites from the impact sites'. No 'gain' coefficient refers to the landscape context. However, it was reported that ecological continuities (*trames vertes et bleues*) were taken into account when identifying potential offsetting sites, but securing such sites was dependent on landowners' willingness. The surrounding environment was nevertheless an element taken into account, with notably the securing of a 113-ha offsetting site bordered by natural areas.

### 3.3. Temporal considerations

**Peru.** In Peru, the offsetting plan must start being implemented when operations are initiated, at the latest (Peru, MINAM 2015). That is, the plan is not executed before the impact occurs. Implementation of the offsetting plan ends when the operator demonstrates that the measurable objectives of said plan have been achieved (Peru, MINAM 2015). One interviewee highlighted that the time lag between the occurrence of the impacts and their effective offsetting is not taken into account in the calculation of biodiversity losses and gains.

Offsetting is, furthermore, required only for non-mitigable impacts. The 2016 offsetting guidelines expressly provide that they do not apply to mitigation plans, nor to the measures or obligations of closure plans and other environmental management instruments. Biodiversity offsetting measures 'are applied in addition to and without prejudice to *in situ* remediation measures, which may be contained in other EIA-d plans' (Peru, MINAM 2015). It was thus reported that offsetting would apply only to cases of irreversible damage (e.g., open pit after mining operations, road infrastructure), as reversible damages are considered at the rehabilitation stage of the mitigation hierarchy. Consequently, if, as part of the operating site's closure, the operator rehabilitates the area, offsets are not required. The absence of an irreversible impact could, as expressed by an interviewee, be a reason for the low number of offsetting plans approved (9, of which 7 in the mining sector) by the competent national environmental authority as of December 2020. Restoration under the mitigation/closure plan is, however, not as demanding as the biodiversity offsetting plan, as the principle of ecological equivalence does not apply. The draft guide for the elaboration of the environmental management strategy provides that the abandonment or closure plan 'aims to restore the areas that were intervened during the execution of the project, in order to reach, if possible, the initial conditions prior to its execution; or to consider the foreseeable future use that will be given to the affected area' (Peru, MINAM 2020).

**France.** In France, weighting equivalence assessment methods can include adjustment coefficients to account for time lags (Andreadakis et al. 2021). In the RCEA case study, one of the 'loss' coefficients applied corresponded to the level of impact, 'based on the level of alteration or destruction of the project (duration and reversibility of the impact) on the natural habitats concerned'. This 'impact level coefficient' is hence meant to account for the time during which the habitat is not suitable for associated species. Further, temporary rights-of-way were assigned a 'relative loss coefficient' inferior to 1 as most will be remediated at the end of the construction phase. As to the estimation of gains, one weighting coefficient is devised to take into account the time lag between the impact and the implementation of the offsetting measure. This 'temporality coefficient' is complementary to the 'habitat dynamics coefficient' also applied, which reflects the time necessary for offsets to be effective. The habitat dynamics coefficient (<1) is meant to account for temporal losses. Ecological variations assessment methods, not used in RCEA, can also include certain weighting criteria in order to integrate temporality issues (Andreadakis et al. 2021). The national method for assessing wetland functions (Gayet et al. 2016a) does not, however, appear to account for temporal losses of ecological functions (Levrel 2020).

### 3.4. Uncertainty considerations

**Peru.** The Peruvian offsetting guidelines do not provide any guidance as to how uncertainties should be addressed. As previously stated, the General Guide and the various complementary guides provide a method to calculate the ecological value of impact and offsetting sites to assess equivalence, but no guidance is provided as to the types of biodiversity offset measures that need to be implemented in order to achieve this equivalence, and there is no indication as to how uncertainty related to offsetting measures should be accounted for.

Under Peruvian law, the environmental competent authorities may, following the result of environmental control actions, require the operator to adopt corrective measures and other types of complementary measures to ensure compliance with the principles and objectives of biodiversity offsetting (Peru, MINAM 2009, 2015). The approved EIA must be updated by the project owner every five years at the latest (Peru, MINAM 2009).

**France.** Weighting equivalence assessment methods can include an adjustment coefficient to account for the effectiveness of offsetting measures (ecological uncertainty) (Andreadakis et al. 2021). In the RCEA case study, a coefficient for qualifying the effectiveness of restoration measures was applied to calculate gains. Its aim is to weight the ecological gain by taking into account the expected effectiveness of the proposed measure (chances of success), on the basis of existing feedback.

Furthermore, under French law, biodiversity offsets are subject to an obligation of results. This obligation is seen as the implementation, or logical continuation, of the NNL objective (Doussan 2018). Corrective measures will thus be required if the monitoring of biodiversity offsets show that the required results are unlikely to be met, provided such monitoring is indeed carried out. The obligation of results applies to the offsetting needs as calculated using the ecological assessment method.

Table 1 below summarizes the comparison of the Peruvian and French policies with respect to the key considerations addressed in this paper.

**Table 1 Comparative summary of similarities and differences between Peruvian and French biodiversity offsetting policies**

Criterion	Peru	France
<b>Ecological</b>	<p>Objective of no net loss of biodiversity and ecosystem functionality, and, if possible, a net gain</p> <p>Principle of ecological equivalence</p> <p>Ecosystem approach required</p> <p>Guidance documents of mandatory application: General guide to the biodiversity offsetting plan, and complementary guides for specific ecosystems (high Andean ecosystems, yunga, bofedales, dry forests)</p> <p>Calculation of ecological value of impact and offsetting areas based on three ecosystem attributes: site floristics, soil stability and biotic integrity. Additional attributes/indicators may be added by petitioners (Peru, MINAM 2016a, b)</p> <p>Attributes may vary depending on the ecosystem considered (see e.g., on bofedales, Peru, MINAM 2019)</p> <p>Guidance on determining the ecological value of an area, but no guidance on how biodiversity offsets are to be implemented</p>	<p>Objective of no net loss of biodiversity, or even a net gain</p> <p>Principle of ecological equivalence</p> <p>No mandatory equivalence assessment method; may be chose freely but must be justified. However, same method must be used to losses and gains</p> <p>Three main categories (Andreadakis et al. 2021): minimum ratio; weighting; ecological variations.</p> <p>=&gt; No uniform method. Can be a combination of different categories of methods</p> <p>Focus on protected species and their habitats, and on wetlands</p> <p>There exists a national wetland functions assessment methodology (Gayet et al. 2016a, b)</p>

<p><b>Spatial</b></p>	<p>Location of offsetting site (choice hierarchy) (Peru, MINAM 2016a)</p> <ol style="list-style-type: none"> <li>1. Within the affected ecosystem</li> <li>2. Within the project's area of direct influence</li> <li>3. In the area closest to the project's area of direct influence</li> </ol> <p>Landscape context mentioned as a criterion to choose offsetting sites (Peru, MINAM 2015)</p> <p>Landscape-related attributes and/or indicators are not always included for the calculation of ecological value (on which equivalence assessment is based). The guide on high Andean ecosystems do not refer to the landscape context, when the guidance document on bofedales mentions landscape alterations as an attribute</p>	<p>Implementation on the damaged site or in its vicinity to guarantee its functionalities (functional and geographical proximity)</p> <p>Additional provision applicable as of December 2022: implemented as a priority within preferential renaturation areas once such areas are identified in land-planning documents</p> <p><i>Ad hoc</i> equivalence assessment method could potentially include landscape-related element for the calculation of ecological gains. They can also include a proximity coefficient reflecting the capacity of colonization from the impact site to the offsetting site</p> <p>Surrounding environment can be taken into account when identifying potential offsetting sites</p>
<p><b>Temporal</b></p>	<p>Offsetting plan to be implemented no later than when operations are initiated. Implementation ends when the operator demonstrates that the measurable objectives of said plan have been achieved (Peru, MINAM 2015)</p> <p>Time lag between occurrence and offsetting of the impacts not taken into account</p> <p>Offsetting required only for non-mitigable impacts. Applies to irreversible damage only</p>	<p>Equivalence assessment methods can include specific coefficients to account for time lags (Andreadakis et al. 2021)</p> <p>Possibility of specific temporality coefficients to calculate biodiversity losses, and specific ones to calculate biodiversity gains</p>
<p><b>Uncertainty</b></p>	<p>No taking into account of uncertainty in the equivalence assessment methods devised at the national level</p> <p>Biodiversity offsetting plan to be updated every 5 years at the latest</p> <p>Environmental competent authorities can require corrective measures (on the basis of the results of control actions – provided such actions are undertaken)</p>	<p>Weighting equivalence assessment methods can include specific coefficients to account for the effectiveness of the proposed offsetting measures.</p> <p>Obligations of results. Possibility of corrective measures.</p>

## 4. Discussion

### 4.1. Addressing ecological considerations

Offsets can rely upon habitat-based (vegetation), species-based (usually fauna), or other calculation methods (e.g., considering alternatives such as ecosystem services) (Bull et al. 2014). Offset practices at international level tend to prioritize rare or short-term threatened biodiversity (Regnery 2017). Of the various equivalence assessment methods reviewed in scientific literature, not many focus on ecosystem functionalities (Parkes et al. 2003; Bezombes et al. 2017; Carreras Gamarra et al. 2018). Assessing ecosystem functionalities is nevertheless encouraged, as it is notably a way to integrate 'common' biodiversity in biodiversity offsets (Regnery et al. 2013; Bezombes et al. 2017). The NNL objective is defined as NNL of biodiversity and ecosystem functionality in Peru, whereas in France it is NNL of biodiversity only. Such a distinction appears to have influenced the approach adopted in each country in relation to ecological equivalence, confirming that the most appropriate metric for a particular context is regulation-dependent (Carreras Gamarra et al. 2018). The Peruvian approach is exclusively habitat-based (quality x area approach), whereas the method used in France will depend on the objectives of the regulation requiring the offsetting (e.g., habitat-based approach for wetlands, and species-based approach under the legislation transposing the European Birds and Habitats directives).

An important distinction between the two countries is also the binding (Peru) vs non-binding (France) nature of assessment methods.

The equivalence assessment method adopted in Peru is an adaptation from the habitat-hectares approach developed in Australia for native vegetation (Parkes et al. 2003). The habitat-hectares method, which is based on compound metrics (Carreras Gamarra et al. 2018), involves comparing existing vegetation features (site conditions and landscape context components) and those of benchmark sites representing the average characteristics of the same vegetation in a mature natural or long-undisturbed condition, providing a global view of habitat quality rather than a species-specific view (Parkes et al. 2003). On the basis of an inventory of existing metrics and corresponding accounting methods, Carreras Gamarra et al. (2018) found that the habitat-hectares approach was the second-best suitable metric for measuring losses and gains, after the Biodiversity Significance Index. Bull et al. (2014) developed a version of the habitat-hectares approach specific to an Uzbek case study, in which the outcome of various equivalence assessment methods was compared. They found that, under their assumptions, the adapted habitat-hectares approach came closest to achieving NNL. The habitat-hectares approach as adapted in Peru is also based on values corresponding to a benchmark area. Given there were no existing high Andean ecosystems in a relatively undisturbed state, the benchmark values were devised using historical information, as recommended in Parkes et al. (2003).

In France, offsets are required for various biodiversity components, depending on the legislation that applies (e.g., protected species and habitats under the –transposed– European Birds and Habitats directives) (Quétier and Lavorel 2011; Bezombes et al. 2018). The logic that prevails, except for wetlands, is based on a protected (or endangered) species approach, with an ecological equivalence based on the functional traits of target species (feeding, conservation and reproduction) (Bezombes et al. 2018; Levrel 2020). It is under these functional traits that the notion of ‘ecological functions affected’, as it appears in the French legislation, is construed in the EIA process. The focus is therefore on the key dimensions of the ecological niche of target species, and not on specific ecosystems. Ecological functions being the biological processes of functioning and maintenance of ecosystems, which include e.g., water retention in soil, recycling of organic matter, regulation of the abundance of organisms (Regnery 2017), limiting them mainly to functional traits of species (for fauna at least) appears restrictive from a legal standpoint, although it may answer the need for operability. Bezombes et al. (2017) found that, in equivalence assessment methods, criteria related to operability are negatively correlated to criteria related to scientific basis and comprehensiveness. In addition, the EIA legislation does not distinguish between so-called common and remarkable biodiversity. The focus should hence not be only on emblematic species, at the risk of not covering all the negative environmental effects of the project (Lucas 2015). Some administrative courts have nevertheless ruled that common biodiversity is taken into account under the habitats entry (habitats of protected species) for both impact assessment and designing offsetting measures, without the need for a specific assessment of common biodiversity (see, e.g., TA Strasbourg 2021). This appears to be a limitation, as there could be cases where protected species and their habitats are not impacted.

#### 4.2 *Addressing spatial considerations*

Spatial considerations include the choice of location of offsetting sites as well as the landscape context. Both countries established rules on the location of these sites, with the idea of proximity to the impact sites. Furthermore, landscape connectivity appears to be a key factor in the success of offsetting (Tarabon et al. 2020; Moulherat et al. 2023), but it is little integrated into the design of offsetting measures (Bigard et al. 2017). Taking landscape connectivity into account could, indeed, help guarantee better biodiversity gains (McKenney and Kiesecker 2010; Quétier and Lavorel 2011; Quétier et al. 2014; Gelot and Bigard 2021).

In Peru, the biodiversity offsetting guidelines mention the need to take into account landscape connectivity. Although the assessment method for high Andean ecosystems is based on the habitat-hectares approach, it does not, however, include any measure that characterizes the corresponding landscape context, unlike the initial approach developed in Australia (Parkes et al. 2003; Quétier and

Lavorel 2011; Carreras Gamarra et al. 2018) and the adjusted habitat-hectares approach developed by Bull et al. (2014) for the Uzbek case study. Other Peruvian guides may nevertheless include landscape-related attributes and indicators, such as the one on *bofedales* which includes 'landscape alterations' as an attribute for assessing EV (Peru, MINAM 2019d). Petitioners also have the possibility to add attributes and indicators, but may choose not to do so in order to avoid additional obligations.

In France, the absence of a nationally imposed method entails that approaches elaborated could be designed to account for landscape connectivity. One commonly used metric is the equivalent connectivity (EC) (Saura et al. 2011), based on the concept of "amount of reachable habitat" for a species or group of species (Saura and de la Fuente 2017). The modeling exercise is not, however, without limitations, which may concern the data used or the number of species targeted (see e.g., Tarabon et al. 2020, 2021; Moulherat et al. 2023). As we have seen in the case of the RCEA, the method applied includes a proximity coefficient to account for the distance between the impact and offset sites, and hence the possibility of colonization from impact to offset sites, but there are no specific coefficient addressing landscape connectivity. However, better gains are likely to be obtained if offsetting sites are close to natural areas (McKenney and Kiesecker 2010; Quétier and Lavorel 2011; Gelot and Bigard 2021), which would allow to stimulate species within offset sites by improving connectivity (Quétier et al. 2014; Tarabon et al. 2021). In our case study, the surroundings of offset sites were taken into account to ensure better gains, in the light notably of the obligation of results imposed on petitioners.

#### 4.3. Addressing time lags

Temporal considerations must be taken into account when assessing equivalence. In particular, equivalence assessment methods must address time lags in order to aim for NNL (Quétier and Lavorel 2011; Maron et al. 2012; Bull et al. 2013; Bezombes et al. 2017). That is, temporal losses must be accounted for. According to Quétier and Lavorel (2011), a multiplier could be applied to a 'quality x area' method (such as the habitat-hectares approach) in order to account for time lags. In Peru, the adapted habitat-hectares approach does not include such a multiplier and, hence, does not take into account temporal losses. Further, as biodiversity offsetting requirements do not concern reversible damage in Peru, NNL and ecological equivalence do not apply. Interim (i.e., non-permanent) losses resulting from such reversible damage are thus not accounted for in Peru, while in France, as seen with the RCEA case study, weighting assessment methods may use a specific multiplier to account for temporal losses, although impacted areas will be rehabilitated at the end of operations. There is, however, no imposed equivalence assessment methods in France; each consultancy can develop its own (Levrel 2020). How temporal losses are taken into account may thus vary. In addition, the national method adopted for wetlands (Gayet et al. 2016a) does not appear to account for temporal losses (Levrel 2020). Assessment methods for wetland offsetting may nevertheless include a time lag factor, such as is the case in the Uniform Mitigation Assessment Method developed by the State of Florida (Quétier and Lavorel 2011; State of Florida nd). Further, the way methods used for *ex-post* ecological compensation account for time lags could be incorporated into *ex-ante* ecological compensation (offsets). As an example, the Habitat Equivalency Analysis (HEA) and Resource Equivalency Analysis (REA) methods, used for *ex-post* ecological compensation, account for delays by calculating gains and losses on an annual basis, and also apply a discounting rate to losses and gains (Quétier and Lavorel 2011). These methods are those advocated under the ELD, which also requires monitoring throughout the period over which gains are calculated to ensure that the gains from the remedial measure cover all the ecological losses associated with the damage (CGDD 2018b). Still in France, two biophysical assessment methods were elaborated to scale ecological damage (*prejudice écologique*) (CGDD 2017, 2018b, c), which include an update factor (*facteur d'actualisation*).

#### 4.4. Addressing uncertainties

Uncertainty of restoration outcomes, which refers to the risk of failure, bears a weight on the effectiveness of biodiversity offsets (Quétier and Lavorel 2011; Maron et al. 2012; Bull et al. 2013; Bezombes et al. 2017). As is the case with time lags, uncertainty considerations must also be incorporated into the design of biodiversity offsets (Maron et al. 2016). Multipliers are commonly used

to account for uncertainties (Bull et al. 2017; McVittie and Faccioli 2020). The application of such multipliers leads to an increase in the compensation area, thus functioning as a precautionary measure (Karlsson and Edvardsson Björnberg 2021). In Peru, the various guidance documents do not expressly take into account uncertainty issues. It could nonetheless be argued that the methods developed to assess the ecological value of specific ecosystems account for uncertainty, as the offsetting surface area will be based on the offsetting site ecological value: the lower the ecological value, the higher the surface area to be covered.

In France, equivalence assessment methods can take into account uncertainty. It was the case in the RCEA case study, where an uncertainty-related coefficient was developed to assess gains. Further, where there is a lack of feedback on certain types of measures, these are not to be considered biodiversity offsets, but support measures (*mesures d'accompagnement*) (CGDD 2018a). Finally, the operator is subject to an obligation of results, which may lead to corrective measures being undertaken in the case of failure. A prerequisite is to ensure that monitoring campaigns are carried out and compliance with operating permits checked.

## 5. Conclusion

Albeit considered unachievable from a strictly ecological standpoint, the objective of NNL underlies most biodiversity offsetting policies, with the corollary principle of ecological equivalence between losses and gains. Biodiversity offsets nevertheless face a number of limitations impairing their effectiveness, which relate to ecological, spatial, temporal and uncertainty considerations. Offsetting being country and regulation-dependent, we compared the approaches adopted in Peru and France to assess whether and how these limitations were addressed, which could inform other countries in their development of offsetting policies as a growing number of States chose to implement such a mechanism, identified as an innovative scheme in the Kunming-Montreal biodiversity global framework (CBD 2022).

Both countries have similar guiding principles in relation to biodiversity offsets. However, Peru has an objective of NNL of biodiversity and ecosystem functionality, whereas France refers only to NNL of biodiversity. Such a distinction has an impact on the approach adopted in each country regarding ecological equivalence. An important distinction is that the equivalence assessment method is binding in Peru, while there is no mandatory method in France. In both countries, we found that the regulation and/or the imposed or accepted assessment methods aimed to address at least some of the limitations identified in the literature in relation to biodiversity offsets. In Peru, the method is based on the Australian habitat-hectares approach and is adapted for specific ecosystems. The landscape context is, however, not necessarily integrated in the method, and temporal losses are not accounted for. Uncertainty is not expressly accounted for either, but can be considered as being indirectly integrated, as a low ecological value of the offsetting site will require increasing its surface area. Further, we lack hindsight and perspective on the implementation of the Peruvian approach as it was adopted recently and only a few offsetting plans using the imposed methodology have been approved thus far.

In France, accounting methods vary greatly, preventing any comparison between offsetting actions carried out for different projects (Levrel 2020). Offsetting in France tend to be mostly focused on protected species and specific ecosystems (wetlands), although the general EIA rules apply to all biodiversity. A more comprehensive interpretation of 'ecological functions affected' (in legislative provisions) should lead to addressing ecosystems more globally, integrating aspects of the ecosystem (e.g., soil) currently left aside. Further, weighting assessment methods may address both time lags (through time discounting) and risk of failure (uncertainty), the national method developed for wetlands does not appear to take such temporal losses into account. As to the landscape context, although necessarily or systematically addressed, it is in the interest of petitioners to take into account the offset site's surrounding areas as it may limit or contribute to meeting the obligation of results.

We hence show that the methods adopted in both countries, albeit different, do not fully address limitations generally associated with biodiversity offsetting, impairing its effectiveness. Our study focuses on limited aspects of offsetting in relation mainly to the design of biodiversity offsets in Peru and

France, in the light of their respective NNL objectives. Other elements would warrant further analysis, in particular monitoring and compliance, without which offset outcomes would remain purely hypothetical.

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### **Electronic supplementary material**

Online Resource 1. Interview guidelines for expert interviews in Peru

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