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The true loss caused by biodiversity offsets

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A B S T R A C T

Biodiversity offsets aim to achieve a "no-net-loss" of biodiversity, ecosystem functions and services due to development. The "no-net-less" objective assumes that the multi-dimensional values of biodiversity in complex ecosystems can be isolated from their spatial, evolutionary, historical, social, and moral context. We examine the irreplaceability of ecosystems, the limits of restoration, and the environmental values that claim to be compensated through ecosystem restoration. We discuss multiple ecological, instrumental, and non-instrumental values of ecosystems that should be considered in offsetting calculations. Considering this range of values, we summarize the multiple ecological, regulatory, and ethical losses that are often dismissed when evaluating offsets and the "no-net-loss" objective. Given the risks that biodiversity offsets pose in bypassing strict regulations, eroding our moral responsibility to protect nature, and embracing misplaced technological optimism relating to ecosystem restoration, we argue that offsets cannot fulfil their promise to resolve the trade-off between development and conservation. If compensation for biodiversity loss is unavoidable, as it may well be, these losses must be made transparent and adequate reparation must embrace socio-ecological uncertainty, for example through a Multi-Criteria Evaluation framework. Above all, strict protection legislation should be strengthened rather than watered down as is the current trend.

1. Introduction

To reach biodiversity protection targets for 2020, the EU will develop [by 2015] an initiative that ensures the "no-net-loss" of ecosystems and their services (e.g. through compensation or offsetting schemes) (EU Commission, 2011 p. 12). As in other parts of the world, ecological compensation via offsets has become a key component of environmental policy. Biodiversity offsets were implemented in the US, France, and Germany in the 1970s, but the policy has recently spread across many countries, accompanied by a convergence of methodology and guidelines. Biodiversity offsets are generally implemented following adherence to the "mitigation hierarchy" of "avoid, minimize, mitigate" within an environmental impact assessment (McKenney and Kiesecker, 2010). The offset involves trading the loss of biodiversity at an "impact site" for a commensurable gain at the "offset site". The biodiversity "gain" is provided via the restoration of degraded habitat, creation of new habitat (we refer to both as "restoration offsets") or the improved protection of threatened habitat (referred to as "averted loss" offsets). Since averted loss offsets do not strictly fulfil the additionality condition of a true "no-net-loss" policy objective (Quétier and Lavorel, 2011; Bull et al., 2012), several offset policies worldwide favour restoration and enhancement over protection, such as wetland mitigation in the US or fish habitat offsets in Canada (Bull et al., 2012; DEFRA, 2013).

Ecosystem restoration aims to accelerate the recovery of ecosystem attributes, such as composition, functionality, structure or resilience, to similar levels in a target (generally near-natural, mature) ecosystem chosen as a suitable reference (SER, 2004). However, early studies on the recovery of mitigation wetlands, following approval of the Clean Water Act in 1974, already reported low success levels in restoring plant cover (Race, 1985). At the time, restoration techniques were experimental, but after 30 years of practice, studies still document impaired biodiversity and functionality in restored ecosystems (e.g. Ballantine and Schneider, 2009; Moreno-Mateos et al., 2012). Data and logistic limitations often restrict these types of analyses to simple metrics of recovery, usually a few functional (e.g., carbon storage, organic matter in soils, denitrification) or compositional indices (e.g., species richness and abundance or cover). Recent work using more sensitive metrics, particularly of community composition and structure, shows that recovery of ecosystems may take centuries or longer, beyond the range of meaningful prediction or policy planning (Maron et al., 2012; Curran et al., 2014). Still worse, if a dynamic

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baseline is used for assessing gains and the assumed rate of background biodiversity loss is high, biodiversity loss can be "locked in" by the offsetting process (Maron et al., in this issue).

Despite these concerns, restoration offsets are being widely adopted (Madsen et al., 2011), accompanied by changes in conservation governance and funding strategies (e.g. Norton and Warburton, 2015). In this paper, we assess offset policy in light of current knowledge of social–ecological complexity and the current state of restoration ecology. We highlight how offsets lead to multiple losses along the different dimensions of value for ecosystems (i.e. ecological, instrumental, and non-instrumental values). After considering the ecological, regulatory, and ethical context of offsets, we argue that no-net-loss is not a progressive step toward no-loss, as the design of offset policies may worsen the present state of biodiversity and existing policies to protect it. Policy-makers must therefore strengthen regulation to prevent loss altogether, and where clearly unavoidable, employ transparent and participative decision-making processes to resolve the associated trade-offs.

2. The uniqueness and complexity of ecosystems

When a biodiversity policy aims at "no-net-loss" of ecosystems (EU Commission, 2011), the potential scope of what is implied is enormous. The term ecosystem encompasses anything from a "pristine" tropical forest in Brazil to an intensive cornfield in Mexico. Specifying which ecosystems are eligible for a "no-net-loss" objective is of paramount importance (Gardner et al., 2013). For this paper, we restrict our scope to ecosystems that have not been subject to recent, radical shifts in their ecological or evolutionary trajectories directly due to human intervention. This includes anything from mature or old-growth forests to well-established, co-evolved cultural ecosystems, like low intensity managed grasslands or coppice woodland. A key premise of our argument is that almost any natural ecosystem, thus defined, is unique due to its social–ecological complexity, and cannot be replaced or perfectly substituted. Its uniqueness emerges from at least three environmental attributes: (i) place-specific environment (*spatiality*), (ii) distinctive history (*historicity*), and (iii) complex ecological processes and interactions (*complexity*; Fig. 1).

In terms of spatiality, the geology, geomorphology, and hydrological dynamics underlying any ecosystem are unique features that will strongly affect the living community. Geology determines the availability of nutrients (e.g. nitrogen or phosphorus) and elemental conditions (e.g. acid, basic, or toxic components). Geomorphology determines whether fine particles (essential to the development of soils) or bare rock develops, influencing the stability of physical structures. Hydrological dynamics determines the availability and form of water resources and, in the case of aquatic ecosystems, affects propagule availability and the distribution of water-borne organisms based on tolerance to flow speed (Hart and Finelli, 1999). The biotic surroundings of a given ecosystem also strongly influence its composition and dynamics, allowing an interchange of species and pathogens, connectedness with larger trophic webs, and so on.

Regarding historicity, a legacy of events, such as fire, colonization, or droughts, makes each natural site historically-specific. A deeper layer of historicity involves coevolutionary trajectories resulting from a combination of altered spatial patterns of habitat, heterogeneous selection pressures, and fluctuating gene flows across a landscape (e.g. the "geographic mosaic theory of coevolution"; Hagen et al., 2012). By abruptly changing these factors, human impacts may unpredictably

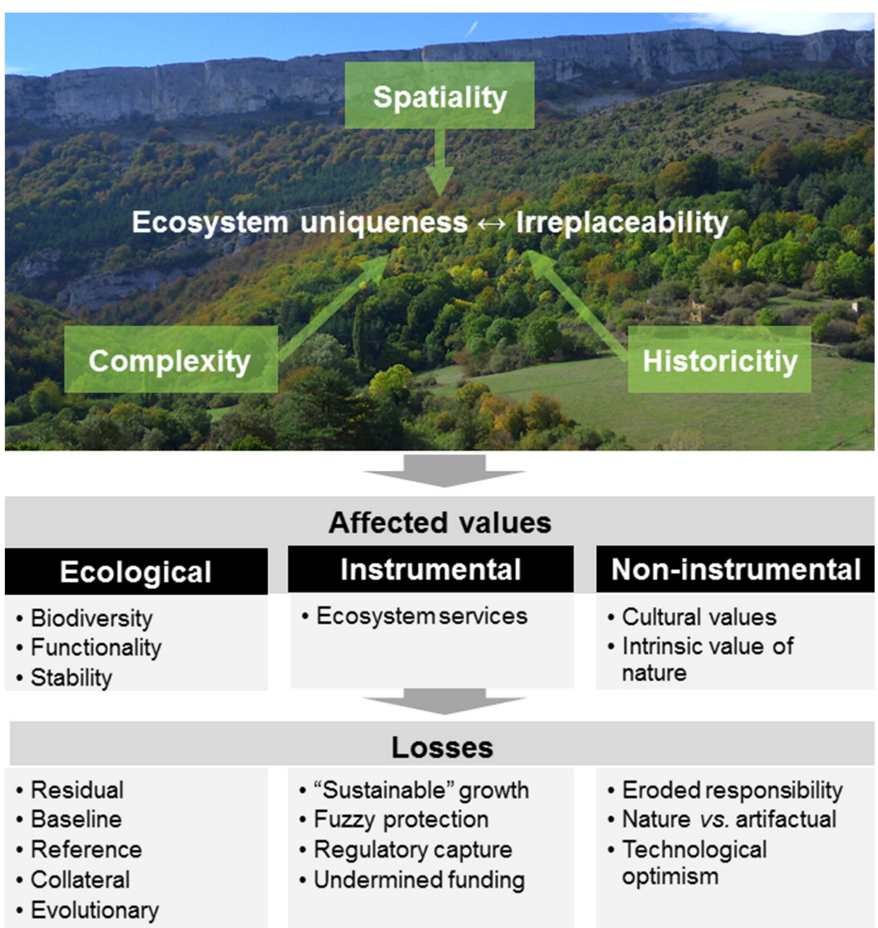

Fig. 1. Losses of ecosystem values caused by biodiversity offsets as a consequence of their irreplaceability.

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alter ecosystem evolutionary trajectories in relatively short time frames (e.g. decades to centuries; Hagen et al., 2012; Zuppinger-Dingley et al., 2014).

Beyond *spatiality* and *historicity*, the *complexity* of interactions among elements in an ecosystem will add to its uniqueness. Complexity arises when systems present emergent properties (Zurek, 1990). Whether ecological systems are truly complex is debatable, as their apparent complexity may ultimately be reducible to mere *complicatedness*, that is to the entanglement of numerous causal links of lower levels without emergent properties (Allen and Starr, 1982). However, from a practical point of view, there is no doubt that many ecosystem-level properties and dynamics appear as emergent, such as top-down regulation by apex predators and the remarkable cascading effects of their loss (Estes et al., 2011).

Taken together, these factors contribute to the appearance of dynamic complexity, characterized by apparent random patterns, sudden discontinuities, self-organization, and regime shifts that are difficult to predict (Hastings and Wysham, 2010). Failure to recognize this complexity is posited to be a major cause of the poor track-record of ecological restoration (Harris and Heathwaite, 2012). Complexity presents potentially insurmountable difficulties in measuring and predicting the outcomes of offsets, which is only partly caused by a lack of empirical data. A more fundamental issue is the epistemological challenge of "knowing" something in the strict scientific sense (i.e. where repeated, direct observation of a system leads to better understanding). Offsets are highly reductionist, applying simple, constructed metrics (e.g. habitat hectares) and generic offset ratios (e.g. DEFRA, 2013). The scientific validity of claims of losses and gains are largely unverifiable, which leads to a perilous "sense of false concreteness". An additional problem raised by dynamic complexity is *strong* uncertainty (e.g. unpredictable sudden shifts or tipping points). While *weak* uncertainty (where a probability distribution can be established or guessed at) can be integrated into offset ratios using info-gap analysis (Moilanen et al., 2009), strong uncertainty cannot and requires precautionary ecosystem management of the whole social–ecological system rather than simplistic accounting of losses and gains at the margin.

These uncertainties are increased by the difficulty to predict social dynamics during offset calculation (e.g. the assumed baseline or background rate of biodiversity loss). For example, in their offset plan for an Ilmenite mining operation in South-Eastern Madagascar, Rio Tinto Inc. assumed a baseline habitat loss rate of ~2%, based on a 10-year national average, to justify the additionality of averted loss and restoration offsets (Temple et al., 2012). By extrapolating this rate to 2065 (the planning horizon), the analysis essentially assumes that Madagascar will remain in an under-developed state, unable to replace wood biomass as an energy source for its impoverished populace, or control illegal deforestation. In essence, such an approach implicitly eliminates hopes of development of the world's poor, and should be a cause of concern from the perspective of social justice and global inequality (Curran et al., 2015). This defeatist "locking in" of loss is also an issue in the developed world, such as Australia's various state-level offset policies (Maron et al., in this issue).

Better monitoring of restoration outcomes and the development of new modelling approaches hold some promise of reducing uncertainty. Approaches combining correlative and mechanistic habitat suitability models should allow better assessment of the impact of habitat loss on population dynamics (Meineri et al., 2015). Other promising ecological approaches include network theory and evolutionary models that integrate ecological, spatial, and genetic networks to derive better metrics of diversity (e.g. food web studies, seed dispersal networks, evolutionary responses), but this still represents an entire field to be developed (Hagen et al., 2012; Montoya et al., 2012; Pocock et al., 2012). Spatially explicit socio-economic models to develop counterfactual scenarios also represent an opportunity for improvement (e.g. Bull et al., 2014), but predictions beyond a decade remain highly uncertain (e.g. for tropical deforestation rates; Brown et al., 2007). In light of all the above discussion, the very notion of achieving "no-net-loss" through restoration (and also largely through averted loss) remains highly questionable.

3. What is not measured, is not compensated

Ecosystem uniqueness and complexity imply fundamental gaps in our understanding of the consequences of human-induced environmental change. As is widely acknowledged in the offset literature, "no-net-loss" does not apply comprehensively to ecosystems, but rather to definable values reflected in offset metrics (Maron et al., 2012; Gardner et al., 2013). For "no-net-loss" to be a useful concept, these values must adequately capture critical attributes of ecosystems (e.g. stability, resilience, resistance, evolutionary potential) and their links to human values (e.g. instrumental and non-instrumental values, including moral considerations of non-human entities; Fig. 1). Identifying these values challenges the origin and limits to contemporary knowledge on biodiversity dynamics, carrying an implicit ethical, cultural, and political perspective on what is important to which social actors.

The value dimension is reflected in the aim of offsets to preserve "components that are particularly valued by people or are of particular functional importance", along with surrogate metrics for "unmeasured biodiversity" (Gardner et al., 2013). The sheer diversity of values affected by offsets is rarely discussed in the offsets literature. Given that failure to achieve no-net-loss is common, even with the conventional ecological metrics, it is important to assess what else is being lost. Indeed, the use of simple, quantitative metrics of ecological equivalence in offsets obscures the diversity and complexity of values at stake, transforming complex socio-ecological patterns and processes, along with their cultural significance, into comodified and substitutable abstract things (Dauguet, in this issue).

3.1. Ecological value

The primary values addressed by biodiversity offsets are biophysical, without reference to any normative content. These values are reflected in simple metrics of quality-weighted ecosystem area (e.g. habitat hectares), suitable habitat area for specific species (e.g. endangered species banking in the USA), and ecosystem functions (e.g. wetland mitigation in the USA; McKenney and Kiesecker, 2010). These and other metrics reflect a small part of local ecosystem composition, structure and function (Bull et al., 2012; Gardner et al., 2013). More complex metrics have been proposed that account for multiple local and landscape-level patterns and processes (Bruggeman and Jones, 2008) or explicitly apply spatial conservation planning tools (Moilanen, 2013), but these also encompass only a small fraction of the overall diversity of an ecosystem. Examples of application include Rio Tinto's "Units of Distribution" (percentage points of threatened species ranges impacted/offset) in their corporate offset policy (e.g. Temple et al., 2012). Offset guidelines in the UK cover a range of ecological components (including connectivity, "distinctiveness", and condition), yet these are reduced to arbitrary multipliers categorically assigned to projects based on ambiguous criteria (DEFRA, 2013). Moreover, the spatial complementarity of sites with respect to the distribution of species and individuals is most often ignored (Kujala et al., in this issue).

More sensitive measures of species composition may better reflect complexity during ecosystem restoration (Jaunatre et al., 2013; Curran et al., 2014), but other proxies are essential. Candidates include: a) the number of interactions among species and between organisms of a same species; b) the strength of these interactions; c) the feedbacks between the abiotic environment (e.g., soil, water and moisture, geomorphological processes) and the biotic community; and d) phylogenetic diversity and evolutionary distinctness at local and landscape scales (Faith et al., 2004).

3.2. Instrumental value

The EU Biodiversity Strategy for 2020 seeks to "ensure no-net-loss of biodiversity and ecosystem services". Ecosystem services are defined according to the Millennium Ecosystem Assessment (2005) as "the benefits people obtain from ecosystems". This definition emphasizes the instrumental value of biodiversity and ecosystem functioning for people. While the ecological values mentioned above are strictly descriptive, the reference to ecosystem services is explicitly normative and can be understood as a way to justify offset policies: if ecosystems provide benefits to people, the destruction of ecosystem should be accompanied by the compensation of the corresponding benefits. Since the links between ecosystem's composition, functions, and human well-being remain difficult to understand and predict, the cautious way to ensure continuing provision of ecosystem services would be to compensate their full ecological value. However, as it is technically difficult, and sometimes impossible, to fully compensate ecological values, offsets may focus on ecosystem services themselves rather than on challenging ecosystem attributes. This shift from ecological values to instrumental values risks loss for at least three reasons.

First, when considered in isolation, ecosystem services are more substitutable than the corresponding provisioning ecosystems. For instance, the regulating service associated to the capacity of ecosystems to store carbon could be compensated or even enhanced by replacing an old growth forest by a more productive planted forest (Liao et al., 2012). Some offsetting policies already target ecosystem services, such as wetland regulation in the French Water Act (LEMA, 2006), requiring developers to offset for the loss of hydrological functionality. Focusing on services could mask losses of other values (e.g. biological entities or ecological functions) potentially leading to conflicts between different types of value.

Second, when considered collectively, it is extremely complex to assess the totality of ecosystem services. This is particularly challenging in the case of cultural services (Milcu et al., 2013) among which the easiest to assess and value (e.g., recreational services or ecotourism) are the easiest to substitute. This can favour some kinds of ecosystem services at the expense of other social values, like sense-of-place or spiritual attachment to nature (Chan et al., 2012).

Third, one single ecosystem can simultaneously provide services and disservices (Zhang et al., 2007) and the same ecological function can simultaneously be a service for some people and a disservice for others (e.g., wetlands that remove nutrients from water and favour mosquito emergence; Knight et al., 2003). This raises distributive justice issues for offsetting schemes to decide who will be the beneficiaries of the targeted ecosystem services and how competing interests are balanced.

3.3. Non-instrumental values

Focusing biodiversity offsets on ecosystem services reduces nature to a benefit provider for humans. However, people's relationships with natural entities are more complex and deeper than this strictly instrumental view (Maris, 2014). Above ecosystem services, two broad categories of values deserve our attention: cultural values and non-anthropocentric values. In the Millennium Ecosystem Assessment (2005), the category "cultural services" includes services characterized by their instrumental value, like ecotourism or recreation, and services that hold values on which the relationship between people and nature are built. For instance, the spiritual values of a sacred site, the cultural heritage associated with specific landscapes, and the sense-of-place that makes people feel attached to their ecological environment, are all cultural values embedded in socio-ecosystems. Aesthetic values lie somewhere between mere services and cultural values. The beauty of a scenery adds recreational worth to some natural environments, but above this utility, beauty and aesthetical emotions are intimately linked to well-being and identity and thus cannot be reduced to their only utility. These cultural values raise two obstacles to biodiversity offsets: first, as intangible goods, they cannot be quantified by ecosystem service valuation. Only qualitative and participative valuation methods can apprehend the complex relationship between people and nature. Second, they are generally connected with history and space. The *de-localization* as well as the artificial *re-creation* of such values are thus problematic, either because the value itself is definitely lost, or because the "compensated" value do not target the same persons or communities.

Beyond the benefits and values for human beings, there are strong arguments for the recognition of the intrinsic values of living beings (Taylor, 1986) or animals (Regan, 1983). These arguments consider them as an end in themselves and not only means for human ends. The destruction of natural habitats can entail the death of large numbers of organisms whose lives cannot be compensated. Whole ecological entities, such as ecosystems or populations, may also be valued independently, given that the moral value of the whole cannot be reduced to the sum of its component values (Leopold, 1949; Callicott, 1989). In *Faking Nature* (1982) Robert Elliot challenges the idea that restored ecosystems could have the same value as natural ones using an analogy with works of art. He argues that faking art would not have the same value as the original masterpiece because it lacks the relational properties linked to the historical context of the original creation. If a special value of natural ecosystems emerges from their autonomy from human intentionality and design, whatever the technical possibilities to recreate an ecosystem are, the restored ecosystem will lack this specific value and will not fully compensate the value of the destroyed one (Katz, 1996).

The different values of ecosystems (ecological, instrumental, cultural, and non-anthropocentric) are neither synergetic nor equally shared by all the stakeholders. Offset design and methodological choices acknowledge which values will be compensated and how, are thus a critical issue for social justice. Such a concern for justice is worthy in and of itself but is also a key factor to ensure the legitimacy and the effectiveness of environmental policies (Pascual et al., 2010).

4. A multiple loss

We acknowledge that it is unrealistic to advocate for the absence of development or transformation of ecosystems. Biodiversity offsets, even if compensating irreversible impacts to ecological, instrumental, and non-instrumental values, could still represent a benefit over a business-as-usual scenario. Despite this, reporting on the limits of "no-net-loss", the multiple losses that are not compensated must be mainstreamed. These and other negative aspects (e.g. perverse incentives) of offset design require a transparency that is currently lacking in offset practice (Gordon et al., 2015). In the next section, we identify multiple losses caused by offsets across ecological, regulatory, and moral dimensions that underpin this concern (Fig. 1).

4.1. The ecological loss

Time, space, and quality imbalances in offset transactions have been extensively discussed in the literature. These include measurement problems, time-lags, risks, and uncertainties, which are compounded at the landscape level due to a piecemeal, project-level focus of offsets to date (Moilanen et al., 2009; Bull et al., 2012; Maron et al., 2012; Gardner et al., 2013). Offset "multipliers" have been proposed to address some issues, but their effectiveness is limited, enforcement is generally weak, and policy recommendations are frequently lower than those recommended by scientists (Moilanen et al., 2009; Bull et al., 2012; Curran et al., 2015). In addition to these issues, we identify further sources of potential loss in offset projects that deserve research attention.

4.1.1. Residual loss

Long-term (from centuries to millennia), direct loss of biodiversity, functions, and other attributes (e.g., stability, resilience, resistance) result from an imbalance between project life spans and actual recovery times for biodiversity and functionality. Life spans of major infrastructure projects, such as bridges, dams, or highways vary between 20 and 100 years, but are usually 50 years or below (Wieland, 2010). In contrast, recovery of degraded ecosystems may take multiple project life spans (Curran et al., 2014). As emphasized in Section 2, our knowledge of restoration and time lags relies on simple metrics, incomplete knowledge, and scarce data, thus even estimating the time lag proves highly problematic. Hence, residual losses (which offsets are designed to address) will accumulate, accentuated by doubts about whether developers can actually commit to finance and monitor offset sites over the relevant time horizons (Rayment et al., 2014).

4.1.2. Baseline loss

An indirect loss emerges from the fact that ecosystems initially devoted to restoration often have a greater biodiversity and functional value than the site resulting from the development project. Consider the case of a semi-natural forest converted into a 1 ha parking lot (i.e. sealed surface), and the offset involves restoring a larger area of an agricultural field back into a forest. An exact exchange would involve reversing development, i.e. require the offset to convert at least 1 ha of sealed surface (e.g., another parking lot) to a similar forest. The agricultural field is likely to have a higher initial biodiversity and ecological functioning than the parking lot and will be much easier to restore (Fig. 2). Offset ratios and multipliers may address the differences in baseline biodiversity, but not the *effort* required to achieve the gain, thus favouring compensation of irreversible losses (e.g. sealing) with easy gains (e.g. enhancement). Restricting trades to like-for-like condition exchanges (i.e. a “condition balance principle”) would be a desirable rule in this context.

4.1.3. Reference loss

For impacts to degraded ecosystems, an “appropriate” reference might be the degraded state (i.e. to match what is impacted), such as young secondary growth forests or hydrologically regulated wetlands. This is beneficial for developers seeking lower costs (i.e. less biodiversity to offset) and cheaper offset credits. Yet this will lead to further intensification and loss of degraded habitats, eliminating the potentially substantial future value of these habitats as they recover (or are restored), and mask the loss of crucial old-growth remnants of the previously intact ecosystem (e.g. old remnant trees; Maron et al., 2010).

4.1.4. Collateral loss

New roads, connections to the electric, gas, water supply networks and sewage system are often required for a given infrastructure development to become operational. It also includes an increase in accessibility, noise, and artificial light that will negatively affect local plant and animal communities and ecosystem functions. These additive effects are, and will likely remain, outside of the scope of offsets (Gardner et al., 2013). For example, prospecting roads for Rio Tinto's Ilmenite mining operation in Madagascar facilitated migration from other regions linked to a massive increase in the deforestation rate (Ingram and Dawson, 2006). These impacts were not considered in the project's offset calculations, despite being of greater magnitude to direct impacts (Curran et al., 2015). The offset may also induce “leakage”, where land set aside for an offset displaces pressures to other areas.

4.1.5. Evolutionary loss

This loss involves the destruction of evolutionary potential for diversification. The evolution of an organism in an ecosystem is shaped by interactions with other organisms (including humans) and the abiotic environment. This process may take just decades to centuries (Hagen et al., 2012; Zuppinger-Dingley et al., 2014), i.e. similar timescales to restoration (Curran et al., 2014), implying evolutionary changes due to an impact may occur despite compensation. Research into the use of metrics of inter and intraspecific genetic diversity (e.g. Faith et al., 2004) in offset equivalence calculations could help address this issue.

Fig. 2. Out-of-kind biodiversity offsetting in Crau (France). Diversity of pre-restoration habitats (top) is higher than the development that is left in perpetuity (middle) and compensated through restoration (bottom). True compensation would aim to renaturate an equivalent artificial area to that of developed, which likely requires much more time and effort than simply enhancing a degraded ecosystem. In this case, there is a net gain of steppe habitat through enhancement but a net loss overall (see Section 3). Photo credit: Axel Wolff — CEN PACA.

4.2. The regulatory loss

Beyond direct and indirect ecological impacts, biodiversity offsets erode conservation policies relying on the strict protection of species and their habitats. This erosion stems from four factors inherent in the design of offset policies.

4.2.1. "Sustainable" growth oxymoron

Offsetting is a slogan used to neutralize the conflict between economic growth and biodiversity protection. The success of the offsetting and no-net-loss rhetoric is partly based on their symbolic strength, because they can be used as a quick-fix to long lasting contradictions in public policies: those encouraging economic growth and those protecting biodiversity (MacDonald, 2010). Ervine (2012) argues that in the case of carbon markets, offsetting legitimises the idea that development should continue without reform, because a new market exists to "fix" the problem. In his view, offsetting is a "psycho-social device" that silences dissent by maintaining the illusion that capitalism is the solution to the problems that it has created. This critique remains valid for the markets of biodiversity offsets.

4.2.2. "Fuzzy" protection

Offsets erode the very meaning of strict protection of species and habitats. By allowing exceptions, offsetting policies state that it is not necessary that every individual of a species or element of a threatened habitat is protected to ensure its persistence. For instance, the Habitat Conservation Plan of the USA's Endangered Species Act authorizes the destruction of members of a threatened species and thus erodes its general rule of absolute prohibition on species take (Ruhl, 1999; Walker et al., 2009). Public belief and trust are shaken by such exceptions as it allows bypassing restrictive regulations, in particular, by those who can afford to pay for it.

4.2.3. Regulatory capture

Commonly, national protection and regulations are downscaled by offsetting policies which generate case-by-case transactions that are under local regulation authorities. NGOs and civil society are often less informed and have less access to regulatory authorities than developers, opening the door for agency capture (Clare and Krogman, 2013). Public choice theory predicts that authorities may act in their own interest rather than in defending the common good (Buchanan, 1984). As such, and in the absence of proper financial and political incentives to guarantee law enforcement, authorities will favour less risky options that satisfy both the developers' requests and compliance to environmental requirements. This alignment of interests between developers and local authorities comes at the expense of the public good, i.e. biodiversity conservation (Walker et al., 2009).

4.2.4. Undermined conservation funding

Offset policies contribute to declining public agency responsibility for funding biodiversity protection. In France, the Ministry of the Environment encourages offsets to support National Action Plans endorsed for several species, but with no baseline funding (Challeat and Lavarde, 2014). For example, the Ocelated lizard *Timon lepidus* in South-east France benefited from €100,000 in offset funding by a motorway company. More generally, offsetting policies displace the burden of funding conservation from regional or state public budgets to developers via offsets associated with infrastructure development. Contrary to the generally long-term visibility given by public agencies to ensure functioning of national parks or nature reserves, offsetting generally provides short term funding. For all these reasons, it is unclear how biodiversity offsets are a gain regarding the business-as-usual scenario and not used as a new bypass to avoid stricter protection of nature.

4.3. The ethical loss

Beyond the axiological issue of the true nature of nature's value evoked in the ethical debate opened by Elliot's essay *Faking nature* (1982), offsetting policies may have a detrimental impact on our moral responsibility toward the protection of natural habitats and biodiversity through three pathways.

4.3.1. Eroded moral responsibility

The option to offset erodes our moral responsibility toward biodiversity protection. A long history of advocacy for nature has reflected the special values of nature, wilderness, and non-human living beings. This wakening of political concern and ethical consciousness has translated into legal and regulatory tools, nationally and internationally. At the national level, the US Wilderness Act of 1964 or the Endangered Species Act of 1973 proclaims such non-anthropocentric commitments toward the preservation of wilderness and the conservation of species. The Bolivian and Ecuadorian constitutions go as far as recognizing the right of nature "to integral respect for its existence and for the maintenance and regeneration of its life cycles, structure, functions, and evolutionary processes"(Asamblea Constituyente, 2008). At the international level, the preamble of the Convention on Biological Diversity recognizes "the intrinsic value of biological diversity" (CBD, 1992). Even if these texts are not binding, they set the baseline about our responsibility toward nature conservation and shake the so-called anthropocentrism that some authors have identified as a root of the present environmental crisis (White, 1967). By institutionalizing the fact that nature can be destroyed as long as the destruction is compensated, we ruin the special sense of responsibility that has been so hard to mainstream.

4.3.2. Nature vs. artefact

For Brennan (1984), the characteristic of natural entities is their lack of intrinsic function. We tend to describe natural processes as if there was an intention at work (e.g., a predator 'plays a role' in an ecosystem). However, natural entities exist without pre-established purpose. This lack of design has been used as a criterion to separate natural entities and processes from artificial ones. Following this distinction, Katz (2000) argues that restored ecosystems are inherently artificial since they are purposefully designed by human beings. Even worse, the motivation for restoration in offsetting policies is not to engage oneself in a positive and reciprocal relationship with nature (benevolent restoration), but to obtain the right to destroy (malevolent restoration; Light, 2000).

4.3.3. Technological optimism

The mere idea of restoration follows the same "technological fix" mentality than some have claimed as responsible of the present ecological crisis (Katz, 1996). Even if the intentions of restoration can be valuable (e.g. biodiversity preservation) and its ecological impacts can sometime be positive (e.g. habitat recovery), ecosytem restoration remains "a continuation of the human project of the domination of the natural world" (Katz, 2012). Confronted by the unwanted effects of the vast technological means to subjugate natural processes (e.g. through pollution, land degradation, climate change), modesty and precaution may be the best tools to cope with the environmental challenges of the future. The ambition to use restoration ecology in order to compensate, and thus to allow, the destruction of the remaining unexploited or less-exploited parcels of nature exalts human pride and reinforces confidence in technological power.

5. Conservation implications

We have argued that the values of ecosystems are heterogeneous, plural, and potentially contentious (i.e. incommensurable). Because of the immense complexity of socio-ecological systems, the direct gains of offset are highly uncertain, and the possible indirect losses (ecological, social, and moral) are too great to adopt without criticism. A large body of work has already investigated the theoretical and practical shortcomings of offsets (e.g. Walker et al., 2009; Bull et al., 2012; Maron et al., 2012; Gardner et al., 2013; Clare and Krogman, 2013; Spash, in this issue) and our work builds upon these efforts to provide a more

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comprehensive account of what is at stake. Considering the multiple dimensions of the problem, achieving "no-net-loss" appears impossible in practice. Proponents of offsets largely accept this fact (e.g. Gardner et al., 2013), and might object to our critique that (1) offsets are "better than nothing" in an era of failing conservation policy, and (2) we do not propose concrete alternatives. We address both points below.

5.1. "Don't let the perfect be the enemy of the good"

A common defence of offsets is that some compensations are better than none. However, this is far too simplistic a view of the social and political context in which offset policies are implemented. First, offset policies arise as both additions and modifications of wider policies, and have led to the weakening of existing, stronger regulation. This includes the mitigation provision under section 404 of the US Clean Water Act (Hough and Robertson, 2009), the "no take" provision of the US Endangered Species Act (Ruhl, 1999) and an offset provision introduced as part of Brazil's revised forest code (Soares-Filho et al., 2014). Granted, these policies encountered compliance issues, resistance from interest groups, and are often perceived as ineffective in addressing biodiversity loss. However, in modifying or improving them there is no a priori reason to favour offsets over other options that may be less fallible to regulatory failure (Walker et al., 2009; Clare and Krogman, 2013; Curran et al., 2015). Perverse incentives linked to offsets mean they may also worsen biodiversity outcomes even if purely additional to existing policies and applied using "best practice" guidelines (Gordon et al., 2015). These include entrenching or worsening biodiversity loss baselines (see Maron et al., in this issue), draining non-offset conservation funding, crowding out other motivations for conservation, and misrepresenting no-net-loss as a "gain" for conservation (Gordon et al., 2015). Finally, the introduction of offset legislation is a policy response to demand for better biodiversity conservation. In their absence, other policies would emerge to meet this demand. Thus, the "offsets or nothing" argument presents a false dichotomy that lacks any counterfactual validity (Curran et al., 2015).

5.2. Suggested alternatives and improvements

Offsets operate in the context of development decisions where facts are highly uncertain; values are plural, contestable and heterogeneous across social actors; and risks are high with potentially irreversible consequences (e.g. loss of old-growth habitat, species or core socio-cultural values). The current emphasis on top-down, expert assessment to make "necessary simplifications" to produce highly contrived metrics of ecological value represents a naïve response to this complexity and uncertainty. One way to tackle both socio-ecological uncertainties and the pluralities of values attached to socio-ecological systems could be to adopt a (Social) Multi-Criteria Evaluation (MCE) framework that combines top-down and bottom-up knowledge through stakeholder participation to arrive at transparent and accountable "compromise solutions" (Martinez-Alier et al., 1998; Munda, 2004). MCE is a decision-support approach to deal with *technical incommensurability* (i.e. measurable properties of a system that can be observed and weakly compared, but not combined in a single metric) and *social incommensurability* (i.e. higher-level perceptions and values that differ across social groups and are influenced by politics, participation, ethics, or power; Munda, 2004). The interaction of the technical and social means expert recommendations cannot be taken at face value, and should undergo an "extended peer review" process with key stakeholders (e.g. focus groups, deliberation workshops). Rather than presenting a single optimized solution, MCE makes conflicts transparent in an attempt to promote compromise through stakeholder exchange and deliberation. It involves using participation to define a set of policy options (e.g. different development or compensation options) ranked along a range of criteria, both informal and expert-based (Martinez-Alier et al., 1998). Such an approach could be integrated into the offset planning framework (or vice-versa), but implies a major expansion of the role of public participation in determining alternative scenarios, identifying values, establishing metrics, setting offset ratios, identifying "no go" options, etc. MCE is also the only one possible approach that should be investigated alongside alternatives.

In essence, our main argument is that, with present knowledge, compensation is not achievable and that using offsets as a trading tool for both tangible and intangible ecosystem values results in the loss of biodiversity and ecosystem benefits to societies. Human agency cannot replace or manipulate nature as a technical exercise through restoration. Thus, the unseen loss will continue for the foreseeable future or until the false goal of no-net-loss is replaced by no-loss. This scenario will be characterized by further destruction of natural habitats, increasing inequity in the distribution of environmental services and values, the strengthening of power asymmetries in development and conservation decisions, and the negation of the intrinsic value of nature. An exception may be when major benefits to society at large (e.g. essential infrastructure or public services) depend on that loss, but demonstrating this requires forms of decision making that fully embrace participation, transparency, fairness and legitimacy.

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