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**Mining and biodiversity offsets: A transparent and science-based approach to measure “no-net-loss”**

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# Abstract

Mining and associated infrastructure developments can present as economic opportunities that are difficult to forego for developing and industrialised countries alike. Almost inevitably, however, they lead to biodiversity loss. This trade-off can be greatest in the poorest developing countries that have highly biodiverse regions. Biodiversity offsets have, therefore, increasingly been promoted as a mechanism to help achieve both the aims of development and biodiversity conservation. Accordingly, this mechanism is emerging as a key tool for multinational mining companies to demonstrate good environmental stewardship. Relying on offsets to achieve “no-net-loss” of biodiversity, however, requires certainty in their ecological integrity where they are used to sanction habitat destruction.

Here, we discuss real-world practices in biodiversity offsetting by assessing how well some leading initiatives internationally integrate critical aspects of biodiversity attributes, net loss accounting and project management. With the aim of improving, rather than merely critiquing the approach, we analyse different aspects of biodiversity offsetting. Further, we analyse the potential pitfalls of developing counterfactual scenarios of biodiversity loss or gains in a project’s absence. In this, we draw on insights from experience with carbon offsetting. This informs our discussion of realistic projections of project effectiveness and permanence of benefits to ensure no net losses, and the risk of displacing, rather than avoiding biodiversity losses (“leakage”). We show that the most prominent existing biodiversity offset initiatives employ broad and somewhat arbitrary parameters to measure habitat value and do not sufficiently consider real-world challenges in compensating losses in an effective and lasting manner. We propose a more transparent and science-based approach, supported with a new formula, to help design biodiversity offsets to realise their potential in enabling more responsible mining that better balances economic development opportunities for mining and biodiversity conservation.

**Keywords**

# Biodiversity Offsets, Biodiversity Accounting, Biodiversity Impact Assessment, Carbon Offsets, Corporate Social Responsibility, Responsible Investment, and Responsible Mining

# 1 Introduction

The perceived conflict between economically profitable ventures such as mining and natural heritage is becoming increasingly acute as the material needs of a growing global population compete with shrinking habitats harbouring the world’s remaining biodiversity. Mining and associated large infrastructure developments create economic opportunities that may seem too significant to forego, even where they threaten to destroy valuable habitat. This is true for developing and industrialised countries alike, although the dilemma can be greatest in the poorest developing countries and regions harbouring globally unique biodiversity, such as shrinking areas of tropical rainforests.

Biodiversity offsets have been promoted as a mechanism that promises the possibility of reconciling both the aims of development and conservation. The mechanism has become a key tool for multinational mining companies to demonstrate good environmental stewardship and manage regulatory risks (BBOP, 2009, 2013; McKenney and Kiesecker, 2010; ICMM and IUCN, 2013). The basic approach is to quantify biodiversity losses that will be caused by development or extension of a mining project that occur even after implementing primary impact mitigation measures, and then to generate biodiversity benefits through compensatory activities that “offset” such residual impacts. Offsetting measures are designed to achieve “no-net-loss” or “net-gain”, compared to a realistic counterfactual scenario (also known as “business-as-usual” scenario) for the site. This could include, for example, protection of comparable forests under threat or active restoration and recreation of habitat that is lost. Offset activities should ideally create or preserve “like-for-like” habitat (ecologically equivalent to the impacted area), although some proponents argue that “out-of-kind” offsets (not ecologically equivalent to the impacted area) can be acceptable where offsets ensure biodiversity conservation of sites with more significant biodiversity components than the impacted area, ideally at the landscape level (Gardner et al., 2013).

The mining industry internationally is spearheading development of biodiversity offsets, not only to conform with national or state regulations, but also to reduce reputational risks and to maintain “social licence to operate” (ICMM and IUCN, 2013). There is also an increasing trend for biodiversity offsets to become an integral part of corporate social responsibility programs. For example, the mining giant Rio Tinto has pledged that its activities will create a “net positive impact” on biodiversity, and offsets are expected to play an important role in meeting this objective (Rio Tinto, 2008). Also several major investors (e.g. International Finance Corporation) have also incorporated “no-net-loss” principles for biodiversity into their investment safeguards policies (ICMM and IUCN, 2013).

Relying on offsets to avoid net biodiversity loss, however, requires a high level of certainty in their ecological integrity where they are used to “sanction” the loss of valuable habitat or rare species. Given the complexity of the concept and the importance of “getting it right” from a scientific perspective, it is not surprising that a number of criticisms have been levelled at biodiversity offsetting. These concerns have included the absence of clear guidance for biodiversity accounting frameworks (Gardner et al., 2013), lack of evidence of actual effectiveness (Gibbons and Lindenmayer, 2007), insufficient governance support to ensure offsets do not undermine crucial prior steps in the mitigation hierarchy, and ensuring risks are fully taken into account (Gardner et al., 2013). Thus, how to ensure that critical factors of risk, effectiveness and permanence, are adequately taken into account in a biodiversity accounting framework, still remains a challenge, and without properly accounting for these parameters, it is difficult to measure the “no-net-loss” principle.

In this paper, we discuss real-world practices in biodiversity offsetting. We do not aim to merely present a critique of shortcomings of current biodiversity offset initiatives, but rather to contribute to a scientifically grounded debate about ways to improve this innovative mechanism while remaining practical within the constraints of real-world application. Our aim is to propose a transparent and science-based approach to measurement of “no-net-loss”.

**1.1 Case study data and methods**

We use real-world practice in biodiversity offsetting from Australia, Madagascar, Ghana and South Africa to assess how well some internationally leading initiatives integrate critical aspects of biodiversity attributes, net loss accounting and project management (Table A.1). In particular, we focus on the case of open-cast mining for ilmenite by Rio Tinto-QMM in the littoral forest, Southeast Madagascar. This case study is critical because it is perceived as one of the first global pioneering and state-of-the-art voluntary biodiversity offset projects that will lead to “no-net-loss” (Temple et al., 2012), despite projections that mining there will destroy more than 50% of the littoral forest on sand, which is arguably a unique and highly biodiverse ecosystem (Ingram et al., 2005; Watson et al., 2005, Virah-Sawmy et al., 2009). We have used a set of published biodiversity data from the southeast littoral forest to compare different methodological approaches for accounting for biodiversity offsetting. We compare outcomes based on using different critical parameters: biodiversity attributes, counterfactual scenarios, leakage risks and effectiveness using a precautionary approach (Figure A.1).

**2 Defining a unit for biodiversity**

Biologists have found it virtually impossible to quantify biodiversity, and no one single measure is ideally suited for all purposes (The Royal Society, 2003; Scholes and Biggs, 2005; Biggs et al., 2005), yet this is specifically what those implementing biodiversity offsets attempt to do. The central importance of defining a “unit” of biodiversity lies in the fact that the composite negative impact of, e.g., a mining project needs to be able to be quantified and compared to the composite benefit of the offset. The use of a large number of key biodiversity attributes and associated surrogate metrics (Table A.1) has been suggested to ensure biodiversity is well represented (see also Gardner et al., 2013). How to define a “unit of biodiversity” is a critical consideration, and we demonstrate below that an evaluation of the effectiveness of an offset can greatly differ depending on which “dimension” of biodiversity is assessed.

## 2.1 Measuring habitat value or condition: The habitat hectare approach

The majority of biodiversity offset schemes to date have used land area as a basic unit of measurement and adjusted this according to a chosen formula to account for different biodiversity attributes such as composition, structure and function of biodiversity and, in some instances, characteristics that operate at a landscape scale, such as connectivity; these are then subjectively weighted into an overall value (BBOP, 2009). Existing biodiversity offsets schemes differ substantially with regard to which biodiversity attributes and weighting system they use (Table A.1).

This approach of combining area and biodiversity attributes is known as the *habitat hectare* and was first introduced in the design of biodiversity offsets in the Australian state of Victoria (Department of Natural Resources and Environment, 2002). It has become a key template that also has been used in most voluntary offset initiatives internationally as well as in Australia (e.g. all pioneering case studies presented in Table A.1 use habitat hectares). This commonly used area-based approach means that both the area affected by a development and the biodiversity protected and/or enhanced by compensatory measures need to be multiplied by some measure of biodiversity value. This is generally expressed as the following but is conceptually flawed as we will show later:

Biodiversity net change: [gain in ha x biodiversity value at offset site – loss in ha x biodiversity value from impacted site] (1)

The habitat hectare approach was developed because it was considered to: be a practical and cost-effective measure, require only relatively simple site measurements, and be easily communicated during engagement with land managers and the broader community (Department of Natural Resources and Environment, 2002). However, the method ignores important aspects of biodiversity. For example, it does not aim to evaluate the conservation significance of a site (e.g. presence of rare or threatened species, or ecological communities), nor does it consider viability requirements of endangered species (see Maron et al., 2010). It has been argued, therefore, that this metric may only be appropriate in habitats with “low” to “moderate” biodiversity values (Gibbons and Lindenmayer, 2007).

Despite these limitations, the habitat hectare approach is commonly used in most voluntary biodiversity offsetting initiatives, including in highly biodiverse areas, e.g., in tropical forests (Table A.1). The question arises then about the range of values obtained via the habitat hectare approach and what can be considered as key biodiversity attributes?

Using existing empirical data for the aforementioned most advanced biodiversity offset project, that of Rio Tinto-QMM in the littoral forest of Madagascar, we demonstrate that measurement of biodiversity attributes via various habitat hectare methods can vary enormously in a landscape (Table A.2). For example, the total habitat hectares impacted could vary from 680 habitat hectares using the Rio Tinto-QMM methodology based on vertical structure of the forest, to 1480 habitat hectares using plant diversity as a measure, or 1,217 habitat hectares impacted, for the average value between plant and bird diversity, or 1,396 habitat hectares using the utilitarian value of plant diversity to local communities (Table A.2, Figure A.2a). The utilitarian value of the landscape to local communities is arguably helpful in achieving effective conservation, considering the opportunity costs to such communities. This is compared to 1,560 hectares of forest affected directly by the mine (Figure A.2a). Thus, our analysis indicates the limitations of relying on any one indicator to determine overall biodiversity value for a particular site.

The habitat hectare approach attempts to deal with different biodiversity attributes by using a weighting system for biodiversity attributes, based on what is considered as contributing most to ecological viability (Table A.1). A weighting system has also being adopted as the standard practice in voluntary biodiversity offset initiatives (Table A.1) and each offsetting case study discussed here has developed and used its own weighting system. But adoption of a single measure in relation to a weighting system to characterise biodiversity can potentially pose a fundamental challenge for objectively calculating the impacts of a development to determine the adequacy of compensatory measures, if measures vary very significantly as we showed in the case above.

We therefore recommend that ecologists working on biodiversity offsets must clearly communicate to policy-makers, industrial stakeholders and civil society that collapsing the multi-faceted dimensions of biodiversity into a single unit will necessarily remain arbitrary and will conflate and obscure the diverging qualities of certain key indices and processes when these measures are decoupled from each other. Instead, both the cumulative sum of net changes of each biodiversity attribute, and as well as net changes for each biodiversity attribute, should clearly be shown in order to indicate whether and where overall net-loss is achieved (Formula 2); that is, was the non-net-loss principle achieved for plant diversity, bird diversity, endangered species and so on?

Importantly, we do not recommend compressing biodiversity attributes into one dimension because it also obscures, and inevitably will lead to very poorly or un-targeted approaches in the selection of the ideal or best management options to produce gains for each of the biodiversity attributes, via compensatory offsets measures, which we will explore later when discussing parameters of effectiveness and permanence (section 3.1).

## Measuring conservation significance: Irreplaceability and vulnerability

Critically, the habitat hectare method is not directed towards evaluation of the conservation significance of a site, such as the importance of the site with regard to the presence of rare or threatened species or vegetation communities. The habitat hectare approach described above is therefore one approach to evaluate the biodiversity of an impacted site. In Australia, the habitat hectare method has been recognised as flawed because it does not take into account conservation significance and there have been attempts to address this through other decision-making aids. For example, “irreplaceability and vulnerability” filters are now applied to land clearing applications in Victoria such that clearing (and hence offsetting) cannot occur in areas containing rare ecological communities that have already been highly cleared (>70%) relative to their estimated pre-European distribution i.e. due to their high conservation value (Gibbons et al., 2009). Nonetheless, this method does not account for determining the significance of threatened or micro endemic species in ecological communities that are above the 70% threshold.

Rio Tinto-QMM has approached the question of conservation significance by developing a novel metric – *units of global distribution* (UD) for high priority species, which are either local endemics or have a conservation status in the IUCN categories of endangered or critically endangered (Temple et al., 2012). A unit of global distribution is described as equivalent to 1% of the total global population of a species (or 1% of its global distribution, in the event that population data are unavailable). Units of global distribution are calculated as follows: if a species has a global population of 1,000 individuals, and 10 of those are destroyed, this would represent a loss of 1% of the global population of that species or 1 UD. Similarly, the destruction of 10 ha of a global habitat of a species of 1,000 ha would represent a loss of 1 UD (Temple et al., 2012, p29). This metric is relatively time-intensive to establish but it has allowed for a better understanding of biodiversity impacts linked to the risk of extinction of specific species than canbe expressed by habitat hectares (Temple et al. 2012), although it assumes that conservation practices will be 100% effective (see later).

A different but less time-intensive way to evaluate irreplaceability and vulnerability of species impacted is to use a *conservation significance index*. This index is measured by determining the number of locally endemic or threatened species relative to the area of remaining habitat (of the species) as a percentage, multiplied by the impacted area or offset area (and applied to formula 2 (see later)). The higher the index, the more significant is the site.

It must be emphasised that the conservation significance index is not a unit but an index that serves to provide guidance regarding the difficulty of offsetting impacts at a site, particularly where endemism is high in the impacted area. For example, as the number of plant micro-endemics in Madagascar’s littoral forests is extremely high and the spatial distribution of the habitat is very small (Rabenantoandro et al. 2007), the conservation significance of impacted forests is well above any chosen out-of-kind offset (Figure A.2d). Accordingly, the conservation significance index could play an important role in evaluating areas where proposed or current offsets include out-of-kind offset habitats.

# Counterfactual scenarios: No-net-loss compared to what?

## 3.1 What area is affected?

Our discussion thus far has focussed on how to define the biodiversity value of a site. The next step is then to calculate whether no-net losses (or even gains) can be achieved from an offset scheme compared to the impact of a mining development on a site. Therefore, a plausible counterfactual scenario needs to be established. If, e.g., forests at a mining site were at the point of being destroyed by other non-mining factors or agents, then it could be argued that the mine creates no net negative impact. Conversely, if habitat at the offset site was under no threat of degradation, then protection of that habitat would achieve no net benefit. As such, counterfactuals play a crucial role in designing compensatory measures and quantifying the actual net benefits they create.

Counterfactual scenarios require a process of making inferences about the likelihood of a hereto-unobserved event: how much future deforestation would be witnessed had there been no conservation (or mining) project? How many lemurs would be have been poached without additional protection? How much passive regeneration would have occurred without active restoration? Answers to these questions will, by their very nature, always be contestable, and there arguably exists a perverse incentive in defining such scenarios if strictly independent actors do not develop them. Nevertheless, any evaluation of the real-world effectiveness of conservation measures – even beyond biodiversity offsets – is incomplete without considering counterfactuals and transparently communicating their rationale in relation to each biodiversity attribute (see formula 2).

The greatest efforts to date in developing counterfactuals have been undertaken in the context of carbon offsets, most notably initiatives for Reducing Emissions from Deforestation and Degradation (REDD) associated with implementation of the UN Framework Convention on Climate Change . Methodologies have been developed for projecting historically observed trends, modelling impacts of independent variables, such as crop prices or population growth, and adjusting counterfactual deforestation rates by comparing them to representative reference regions outside of the reach of project impacts (Huettner et al., 2009).

To illustrate the complexity and care that is needed with counterfactuals, we revisit the Rio Tinto-QMM project. The company makes two fundamental assumptions, as mentioned above: 1) the vertical structure of the forest represents the overall biodiversity value of the habitat, thus 680 habitat hectares was estimated as the unit of biodiversity lost; and 2) given on-going deforestation pressures in the region, much of the littoral forest would have been lost. This assumes a conservative national deforestation rate of 0.9% to establish the project’s actual net impact (Temple et al., 2012). Using such counterfactual scenario means net mining impact is reduced to 313 habitat hectares loss over 60 years of mining operations (Figure A.2a). Therefore, depending on the biodiversity attribute and the counterfactual used, the amount to compensate can vary from nil (at a predicted rate of 4% background deforestation using any of the biodiversity attributes) to 1500 habitat hectares (at a predicted rate of zero background deforestation for plant diversity as the biodiversity attribute) (Figure A.2a).

Thus, counterfactual scenario selection is of central importance in estimating impacts. In the Rio Tinto–QMM case, the evidence also suggests a more complex picture. Prior to the planning of mining activities, a relatively stable communal property regime and other conditions enabled a sustainable use and management of the common-property forests in the area for many centuries (Virah-Sawmy et al., 2009). However, preparation for mining development in the region, including the building of roads, since the 1990s may have altered property rights regimes and attracted migrants (Ingram and Dawson, 2006). Newly arrived migrants are known, in fact, to be responsible for a large part of the recent deforestation observed in the area (Ingram and Dawson, 2006). Accordingly, the deforestation rates for the three mining sites within the littoral forests jumped from 0.3% between 1984 to 1999, to a combined 3.3% between 1999 and 2005 (Ingram and Dawson, 2006). By using a background deforestation rate of 0.9%, which the company considers as conservative, it nevertheless appears that some indirect impacts associated with the mining project itself were used in establishing an inappropriately high rate of biodiversity loss in the counterfactual scenario – calling into question the quantification of net losses that would need to be offset.

## How much biodiversity is affected?

The conversion of one hectare of forest to mining use does not necessarily translate into the same biodiversity loss as the loss of one hectare under the counterfactual scenario for the impacted or offset site. For example, the conversion of forest to a landscape of small-holder agriculture may retain significant biodiversity values. That is, one hectare of “high quality” forest lost to traditional agriculture versus a hectare lost to mining may not compare equally.

In REDD projects, this is known as the “emissions factor” (Olander and Ebeling, 2011). For example, not all carbon stored in an area is lost when humid forest is converted to shifting agriculture (a type of agriculture that relies on clearing vegetation, either forest or fallow land, for cultivation, which is then abandoned when fertility is reduced), particularly where long fallow periods are involved. Similarly, carbon sequestered in a forest plantation must be compared to that stored in shrublands, if that was the pre-project vegetation, or may even have to be compared to a counterfactual of passively regenerating forest land (Olander and Ebeling, 2011).

This same basic logic could be applied to biodiversity offsets with introduction of a “biodiversity conversion factor” (see formula 2). Often, the relationship between habitat area converted and net biodiversity losses is both complex and poorly understood, given the many aspects of biodiversity and their unique characteristics in a landscape – a fundamental difference to the more one-dimensional nature of carbon. This makes it tempting to ignore – yet it suggests added research effort is even more urgently required.

The Rio Tinto-QMM project and the Ambatovy offset projects in Madagascar (Table A.1) illustrate this challenge. Both assume shifting agriculture to be one of the main threats to biodiversity, and most of the compensatory offset activities aim at reducing this threat. By simply applying projected deforestation rates (measuring area), both assume that shifting agriculture is comparable to mining in its biodiversity impacts. But the impacts of shifting agriculture can greatly vary to large-scale clearing for monocultures to small clearances for agro-forestry. But in large-scale open-cast mining, for example, the loss of biodiversity in an area is complete, without any time-lags between gradual loss and future recovery. At a minimum, the habitat hectare approach used to assess original biodiversity values on a site could be applied to better take into account relative losses. As discussed above, this approach needs to be adjusted to reliably capture different key biodiversity attributes that may be affected in different ways. For example, in the case of Rio Tinto-QMM, the charcoal exploitation that is prevented at their offset site *could* be considered as affecting only 30% of the impact of mining on bird diversity per habitat hectare rather than 100% because traditional charcoal exploitation causes mainly habitat degradation rather than complete deforestation. In summary, a “biodiversity conversion factor” needs to be applied in the counterfactuals at both the mining and offset site for each biodiversity attribute (see formula 2).

# 4 Effectiveness of biodiversity conservation/interventions

Determining a plausible range of counterfactual scenarios for different biodiversity attributes should provide a picture of the maximum attainable benefits in an offsite project area where compensatory “offset” activities are implemented. If, for example, counterfactual deforestation was optimally controlled, then the intervention would prevent 100% of the counterfactual habitat loss. In reality, however, implementation success will rarely be 100%.

However, the biodiversity offset case studies (Table A.1) assessed here do not implicitly consider the degree of implementation success that may be expected. Instead, the proponents of these projects assume an effectiveness of 100% in preventing biodiversity losses compared to the counterfactual scenario. Arguably it is not possible to forecast the future success of an intervention with a very high level of confidence, especially in a complex biodiversity setting. As such, it would be appropriate to monitor actual performance on an ongoing basis and adjust interventions, including the extent of the area under management, to verify or otherwise a previously projected result.

During offset project design, it is critical that assumptions and key variables concerning assumed effectiveness of different management approaches, e.g. protection from grazing or shifting agriculture through community managed protected areas are clearly specified (see formula 2) and a monitoring plan for the targeted biodiversity attributes is developed. Thus, as we have discussed, it is critical that biodiversity attributes are not always collapsed into one single unit as biodiversity attributes will differ in response to different management practices, and different expectations of effectiveness may need to be applied to different biodiversity attributes i.e. a precautionary approach should be adopted.

In general, offset plans should assume a sub-optimal success rate and compensate this, for example, through a larger area of habitat being put under protection. If, for example, an offset program needed to achieve a reduction of counterfactual deforestation by 1,000 ha, then it may be prudent as a precautionary approach, to target an area of, say, 2,000 ha and assume a 50% conservation success rate, depending on implementation, complexity and degrees of risk that are deemed necessary. The same approach could be used for other biodiversity attributes, e.g. population of a species. When, for example, dealing with the uncertainty of recovery of an endangered species, it may be best to apply an even more precautionary approach to effectiveness.

# 5 Displacement of biodiversity loss: Leakage

A related challenge can arise when a project intervention is simply displacing the problem to another location, outside the managed area. This phenomenon is termed “leakage” in carbon offset standards and methodologies (Aukland et al., 2003) and, although it can partly or fully negate net biodiversity benefits of an intervention, is rarely considered in conventional conservation efforts. Also, it is almost entirely missing in biodiversity offset initiatives to date – a severe shortcoming, in our view. Leakage risks apply both to the offset site and the site impacted by the development (e.g. a mine) itself.

For example, the Rio Tinto-QMM project assumes that much of the area impacted by its mine would have been lost to charcoal production, as well as shifting agriculture, in its counterfactual scenario. Since the project does not create an alternative to shifting agriculture (though it does for charcoal production through fuel wood plantations), it should conservatively assume that shifting agriculture will be displaced and accelerate deforestation in nearby remaining forests. In some respects, thus, applying a counterfactual scenario to a mine site is unnecessary because impacts are often displaced off-site, which becomes obvious when a leakage factor is included in the equation (Formula 2).

In well-designed REDD projects, leakage risks have to be systematically assessed through a causal framework; risks are then mitigated and remaining leakage is quantified to the extent possible and conservatively deducted from project benefits (Ebeling and Olander, 2011). To date, methodologies have been approved for carbon accounting under the Verified Carbon Standard (VCS) (VCS, 2013) and for biodiversity impact assessment under the Climate, Community and Biodiversity (CCB) Standards (CCB, 2013). These standards involve validation and verification by independent approved auditors. We propose that biodiversity offset scheme management could draw on these approaches to ensure no net losses are indeed created (see formula 2).

Consideration of leakage risks can greatly improve project design, for example by assessing whether social compensation measures are likely to sufficiently address costs incurred to communities by restriction of grazing, charcoal production and agricultural activities or access to restoration areas, because of mining or offset activities. In many cases, it may be best to introduce alternative, sustainable practices, for example through bio-energy plantations for charcoal or improved agriculture at the appropriate intensity and scale that is needed to prevent leakage first at the mine site. Then, the basic question for adequate biodiversity accounting becomes consideration of whether offset compensatory measures address the root causes of biodiversity loss so as to minimise the risk of simply displacing the problem off-site again.

# 6 Permanence of biodiversity benefits: No net loss for how long?

The impacts of a mining or infrastructure project can be considered permanent. This implies that compensatory offset measures must create permanent benefits. If they are not, for example if a forest restoration project fails or a conserved forest is cut down in the future, then net losses occur and on-site impacts have not been offset. How best to address this risk is not usually considered in existing biodiversity offset initiatives.

“Non-permanence risks” have been at the centre of criticisms levelled at forest carbon offsets (Marland et al., 2001), and several approaches have been devised to mitigate these. Afforestation and reforestation projects under the Clean Development Mechanism (CDM) of the Kyoto Protocol are only issued with temporary carbon offset credits which have to be renewed or replaced periodically. In addition, afforestation and reforestation and REDD projects need to set aside a portion of carbon offsets into a pooled non-permanence risk buffer under the Verified Carbon Standard. This portion depends on initial and periodic risk assessments. Other carbon standards have trialled insurance schemes or legal permanent conservation obligations (Ebeling and Vallejo, 2011). For all, the underlying principle is a regular independent audit that confirms that the benefit still exists, alongside a mechanism for replacing units that are lost through reversal of benefits.

In the case of biodiversity offsets, it may be possible to develop a risk-buffer system based on habitat hectares. More appropriate, however, may be an enhanced emphasis on long-term effectiveness of interventions, especially through tackling the underlying drivers of biodiversity threats, combined with a “generous” offset design incorporating ample error margins to allow for discounts. Sustained management capacity potentially could be supported through permanent not-for-profit foundations to fund ongoing activities.

Thoroughly assessing both leakage and non-permanence risks, and designing measures based on a transparent causal approach, could greatly improve overall offset project design and effectiveness. Both leakage and non-permanence aspects need to be urgently considered for biodiversity offsets to ensure intended additional and long-term net benefits are indeed created.

# 7 Discussion

Widely accepted aspects of good practice for biodiversity offsets, defined by BBOP (2013) include considerations of: 1) equity; 2) irreplaceability and vulnerability; 3) long-term conservation outcomes; 4) transparency; and 5) good science. Our analysis based on comprehensive ecological science and real-world experience from the conceptually similar field of carbon offsets indicates that the former principles are not well incorporated in several existing and highly promoted voluntary biodiversity offset initiatives because of inherent flaws in biodiversity accounting methodologies and offset design approaches.

Biodiversity offsets share many conceptual features with forest carbon credits in terms of non-project counterfactuals, leakage and permanence risks. For example, afforestation projects yield useful insights for biodiversity restoration, and REDD projects do so for biodiversity conservation. However, important differences exist, mainly due to the fact that carbon is a relatively easily comparable and exchangeable metric (measured in tonnes of CO2 equivalent). Although it may be tempting to also create a “one-dimensional” biodiversity metric, our analysis in this paper demonstrates that a variety of biodiversity attributes need to be considered to capture the much more complex nature of biodiversity, particularly in highly biodiverse regions. This is important not only because key components of biodiversity can be de-coupled from each other but also because parameters such as counterfactual scenarios, realistic and precautionary approaches to effectiveness of compensatory measures, risks of leakage (displacement) of pressures, and permanence of benefits can differ significantly for different biodiversity attributes.

It is therefore critical that the design of biodiversity offset measures and their accounting approach is transparently laid out and scientifically grounded. Describing how all relevant parameters are rigorously considered in a project, and for each of the key biodiversity attributes in a landscape, is critical for building confidence in the concept of biodiversity offsets itself and, by implication, in the sincerity of environmental stewardship aims by mining companies and other actors involved in implementation of offsets.

Based on the considerations in this paper, we propose a comprehensive formula that incorporates several critical parameters and allows for a scientifically robust manner to calculate net biodiversity impacts (Figure A.1). This allows for a transparent estimate of whether no-net-loss is, in fact, achieved by a biodiversity offset programme. It is expressed as follows for compensatory activities at offset site to averting risks to biodiversity loss (see Figure A.1), and also for positive interventions to enhance biodiversity (e.g. restoration):

Biodiversity net impact: (2)

where

*c* is the value of a given biodiversity attribute of 1 to n, or the overall conservation significance index

*a* is the offset site/s, and

*b* is the impacted site/s

ai = ((counterfactual scenario x conversion factor) x % effectiveness x (1 - % leakage))c x (offset area x value c)c

bi = (counterfactual scenario x (1 - % leakage))c x ( impacted area x value c)

We provide an example of how this formula can be applied in practice by using the Rio Tinto-QMM case in the littoral forest of Madagascar. The results indicated that whether Rio Tinto-QMM is predicted to achieve mining with no-net biodiversity loss in the next 60 years of mining is primarily dependent the parameters used from formula 2. Under a low projected deforestation rate counterfactual scenario, there is no net gain for the Rio Tinto-QMM project based on any of the biodiversity attributes, but this changes as the projected deforestation rate is increased to above 0.3% (based on Rio Tinto–QMM vertical structure methodology) to 0.6 % (based on the average diversity methodology estimated from plant, utilitarian and diversity (Figure A.2b). But this scenario has the assumption that the offset project will be able to achieve zero deforestation (100% effectiveness and that there is no leakage, including in mining site), and the conversion ratio is 100%, that is, shifting agriculture has the same impact that mining per area or habitat hectare affected – values regarding these parameters that are unlikely outcomes and risky.

In the Rio Tinto case, a more plausible scenario of 1% annual deforestation in the out-of-kind offset (based on Vielledent et al., 2013) and 0.3% annual deforestation on-sites pre-mining (based on Ingram and Dawson, 2006) should be used to estimate net changes. In addition, estimate of effectiveness including leakage and conversion factor should be applied on specific biodiversity attribute such as plant diversity, utilitarian value, or bird diversity. Using, for example, a very optimistic scenario, of 50% effectiveness rate that would have included leakage at offset site, 0.3% leakage at mining site, and a 70% conversion factor on all biodiversity attributes (though we are fully aware that these parameters may differ depending on which biodiversity attribute is used), our analysis indicates that there is net loss predicted in the next 60 years when more direct biodiversity attributes, such as plant and bird diversity are used, rather than indirect indicators such as forest structure (Figure A.2c). Thus, net-loss is achieved in spite of what can be seen as a “generous” offset area of about 6687 forest hectares to 1665 forest hectares impacted (ratio of 3: 1, also known as a multiplier). Here, we caution against using land-based multipliers as an indicator of predicted no-net-loss because such multipliers do not implicitly indicate anything about net changes in biodiversity attributes as a result of mining and offset activities.

Furthermore, in the case we have analysed, if a conservation significance index is applied, it appears that it will not be possible for Rio Tinto-QMM to offset its impact on a highly endangered habitat (Figure A.2d). This is because none of the out-of-kind offsets have as high conservation significance as the littoral forest mined. We strongly encourage Rio Tinto-QMM and other implementers of biodiversity offset initiatives to re-evaluate their offset methodology/ approach transparently describing parameters used in formula 2.

Ensuring that permanence risks are taken into account for the remaining littoral forest in Madagascar is also critical. This factor cannot however often be “measured” but could be demonstrated by, for example, setting up a foundation to ensure local communities can manage legally protected areas, for perpetuity. A larger out-of-kind offset may also be necessary to create overall net gain if it is accepted that that biodiversity is exchangeable.

**8 Conclusions**

Given the magnitude of global growth in demand for mineral resources and plans for large-scale mining and associated infrastructure developments, the role of biodiversity offsets looks set to increase. If designed and implemented well, they have the potential to help balance economic development with more responsible environmental stewardship. However, a more transparent and science-based approach is needed if this innovative mechanism is to realise its full potential in working towards a more environmentally responsible approach to mining and infrastructure development.

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Tables and Figures

Table A.1: Comparison of metrics and methodologies in regulatory settings in Australia with new voluntary approaches at the global level (most advanced to least advanced from left to right) (italised are comments by the authors, and non-italicised are descriptions of the approaches used in the case studies)

Table A.2: Broad range of values in habitat hectares depending on which biodiversity indices used – case of Rio Tinto-QMM open-cast dredging in the littoral forest

Figure A.1: A generalized conceptual framework of biodiversity accounting for achieving no-net-loss of biodiversity

Figure A.2: Modelling losses against gains using a range of parameters for the case study from Rio Tinto-QMM. The models show that the results are dependent on the biodiversity attributes used, as well as the parameters used including the counterfactual scenario, % effectiveness, and conversion factor. In more details: a) habitat hectare lost based on 3 key biodiversity indicators (indirect indicator based on forest structure using Rio Tinto-QMM methodology, forest hectare loss, and direct indicator based on average diversity estimated from plant, bird, and utilitarian diversity; b) net changes in all offsets using a realistic counterfactual scenario, thus assuming counterfactual scenario of 1% deforestation loss at offset site, and 0.3% at mine site, and assuming 50% effectiveness including leakage and 70% conversion factor at offset sites, and 30% leakage at mine site. Note that restoration was not taken into account in net changes estimation given high uncertainly on resulting biodiversity attributes and the small area concerned, but by contrast, conservation zones in mining perimeter identified to be conserved during mitigation hierarchy, were considered as 100% gained by Rio Tinto, that is the counterfactual scenario is that all would have been lost by mining, which greatly increased the habitat hectare gained by Rio Tinto-QMM even prior to offset activities; d) using conservation significance index as the biodiversity indicator.