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Technical conditions for positive outcomes from biodiversity offsets

An input paper for the IUCN Technical Study Group on Biodiversity Offsets

THE BIODIVERSITY CONSULTANCY

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1 Non-technical summary

We assess the conditions under which biodiversity offsets may: (i) provide the best outcomes for biodiversity; and (ii) achieve no net loss. Here, we consider 'biodiversity offsets' as measurable conservation outcomes resulting from actions designed to compensate for significant residual adverse biodiversity impacts arising from project development after other appropriate prevention and mitigation measures have been taken (BBOP, 2012a). The goal of biodiversity offsets is to achieve no net loss (or net gain) in biodiversity. 'No net loss' is a goal in which residual impacts on biodiversity (after other mitigation measures have been taken) do not exceed the gains from offsets. Importantly, no net loss has no universal definition: it can have varying definitions dependent on what biodiversity and human preferences are accounted for, and how they are accounted for. For example, no net loss goals may vary in terms of spatial scales, biodiversity that is included, or whether they include only 'like-for-like' exchanges (e.g. replacing a hectare of house mouse habitat with a hectare of house mouse habitat) or also 'trading up' (e.g. replacing a hectare of house mouse habitat with a hectare of panda habitat).

How much uncertainty is there?

It has been suggested that uncertainty is hindering uptake of biodiversity offsets (ICMM, 2005a, 2005b). In the last five years, however, much has changed. First, there has been concerted research by the Business and Biodiversity Offsets Programme (BBOP), including development of the first best-practice standard for voluntary offsets (BBOP, 2012a). Second, lessons have been learned from implementation of both voluntary and regulatory offsets around the world. We assess these scientific and practical advances in this report. We conclude that ***the scientific community, informed stakeholders and much government policy have reached consensus on the basic, high-level principles to achieve no net loss and optimal offsetting***. Others have reached similar conclusions – for example, Bull et al. (2013) identified only a few remaining issues that need to be addressed, noting that "….to ensure robust offsetting, research is required on three issues…". However, the consensus principles necessarily remain broad because "there is no unique 'right answer' or formula for what a biodiversity offset should comprise" owing to varying local context (BBOP, 2012a). Underneath these high-level principles, the novel field of biodiversity offsetting would certainly benefit from both more theoretical research and implementation data. Despite high-level consensus, two major additionality issues are identified here as needing concerted further research: (i) definition of additionality of offset-type actions within protected areas and (ii) identification of conditions for ensuring credit-stacking provides incentives for biodiversity but does not allow 'double-dipping' (Section 5.2.8). Other major issues also need resolution (e.g. methods for 'trading up'), but are not obstacles to basic offset systems.

How should we deal with remaining uncertainty?

Consideration in this report of advances in offset science and practical offset implementation demonstrate that ***there is often a trade-off between certainty and simplicity***: basing an offsetting system on the best available science may make it impossible to implement because of the transaction costs such a system would incur. We thus believe, on the basis of this review, that the conservation community should often applaud voluntary offset efforts, actively support attempts to achieve no net loss through best-practice offsets, and provide practical guidance and constructive criticism within a safe learning environment.

How could offset outcomes be most improved?

Current offsetting outcomes would be most improved through ***integration of societal biodiversity conservation goals, greater adherence to the mitigation hierarchy and better implementation***. The overriding reason for high levels of offset underachievement or failure in regulatory systems is ineffective implementation. There is a lack of incentives for effective implementation, owing to limited capacity in authorities for monitoring, oversight and enforcement – a capacity gap which is reinforced by disincentives for strong regulation (Section 5.3.2). In addition to better implementation, three offset design factors would also significantly improve offsetting outcomes. First, offsetting targets and no net loss definitions should be linked to biodiversity conservation goals (and preferably development goals) in separate policies and plans (such as national or provincial systematic conservation plans). Second, greater attention to feasibility testing and stakeholder engagement during the offset design process is likely to result in offset designs that are more practical to implement. Third, in almost all cases, the use of inappropriate offsets can be reduced by greater adherence to the rest of the mitigation hierarchy before jumping to use of offsets.

Under what conditions do biodiversity offset approaches provide positive outcomes for biodiversity, irrespective of the concept of no net loss?

Here we identify the conditions that facilitate better outcomes for biodiversity, from some form of scientifically-based approach to compensate for impacts, compared with the status quo in which compensation for residual losses of development projects is usually absent or inadequate. While the status quo in some countries is much better than this, the lack of compensation for development impacts on biodiversity is unfortunately common across much of the world. We conclude that ***in many cases, even low-quality, incomplete, impermanent, poorly implemented biodiversity offset approaches could provide more positive outcomes for biodiversity than a status quo of limited or inadequate compensation***. There are two main reasons for this: (i) offsets would provide better compensation for impacts than limited or inadequate compensation; and (ii) offsets would provide an opportunity for better outcomes in terms of management of existing biodiversity that is suffering ongoing loss (e.g. threatened species and declining habitats).

Nonetheless, we also conclude that, ***in some cases, offsetting may not improve a status quo of limited or inadequate compensation***. In cases where existing biodiversity protection is already strong, acceptance of low quality offsetting might degrade existing legislation (i.e. reduce existing protection). In cases where there is strong stakeholder engagement in the development process, acceptance of low quality offsetting would facilitate more development consent than may otherwise have taken place, owing to a perception that impacts were being adequately compensated for by offsets. In cases where adherence to the rest of the mitigation hierarchy is currently strong, increased acceptance of offsetting may reduce emphasis on the most important early steps of the mitigation hierarchy (avoidance, minimization) rather than the later steps (restoration, offsetting). In any of these cases, offsetting could potentially lead to more negative outcomes than the status quo. In such cases, conservationists may best enter dialogue with government to ensure that either such policies are not put in place or that pragmatic best-practice policies are put in place and that they are well implemented, monitored and enforced.

Under what conditions is it possible to achieve no net loss through the implementation of biodiversity offsets?

Achievement of no net loss is a much more ambitious goal than that addressed in the first question, and thus requires a much more narrow and strict set of conditions – both technically and politically. We conclude that ***achievement of 'no net loss' is likely to prove challenging, and will require numerous facilitating, technical and implementation conditions to align***. Ecologically defensible design that can be reasonably expected to achieve no net loss may result in requirements that are unachievable (e.g. requiring restoration so advanced that techniques do not currently exist), offsetting systems that are too complex to be functional (i.e. to allow transactions), or costs that are too high to make development profitable (e.g. owing to high multipliers for uncertainty). Success in achieving no net loss is generally more likely where development has relatively small spatial footprint impacts but high economic profit margins that can fund good adherence to the mitigation hierarchy including best-practice offsetting (e.g. much – though certainly not all – of the mining, oil and gas, manufacturing and service industries), rather than extensive spatial footprint impacts and lower economic profit margins (e.g. much agriculture).

2 Conclusions and technical summary

We assess the conditions under which biodiversity offsets benefit biodiversity and achieve no net loss, asking two key questions:

i. Under what conditions do biodiversity offset approaches provide positive outcomes for biodiversity, irrespective of the concept of no net loss?

ii. Under what conditions is it possible to achieve no net loss through the implementation of biodiversity offsets?

There is high-level consensus on offsetting principles.

Overall, there is much consensus (within the scientific community, among informed stakeholders and in much government policy) on the basic principles to achieve no net loss and optimal offsetting. However, these principles necessarily remain generalizations and the novel field of biodiversity offsetting would certainly benefit from both more theoretical research and implementation data to further develop these principles. As other authors have noted, “there is no unique ‘right answer’ or formula for what a biodiversity offset should comprise” (BBOP, 2012a) and thus “detailed guidance remains elusive. In part [reflecting] the difficulties associated with providing one-size-fits-all guidance for offset programs aimed at addressing complex impacts that vary with the local context” (McKenney & Kiesecker, 2010). Nonetheless, two particularly complex additionality issues were identified in this report as needing considerable further stakeholder engagement: (i) definition of additionality of offset-type actions within protected areas and (ii) identification of conditions for ensuring credit-stacking provides incentives for biodiversity but does not allow ‘double-dipping’ (Section 5.2.8). Other major issues also need resolution (e.g. methods for ‘trading up’), but are not obstacles to basic offset systems.

Achievement of ‘no net loss’ is likely to prove challenging.

From a conservation perspective, ‘no net loss’ – i.e. full compensation for negative impacts of development – is an admirable and sensible goal. In practice, incorporation of all externalities, in order to achieve no net loss, has proven difficult. Ecologically defensible design that can reasonably be expected to achieve no net loss may result in requirements that are unachievable (e.g. requiring restoration so advanced that techniques do not currently exist), offsetting systems that are too complex to be functional (i.e. to allow transactions), or costs that are too high to make development profitable (e.g. owing to high multipliers for uncertainty: Moilanen et al., 2009). This should come as no surprise, except to those who believe ‘win-win’ situations are anything but a rarity when it comes to reconciling conservation and development concerns. Success in achieving no net loss is generally more likely where development has relatively small spatial footprint impacts but high economic profit margins (e.g. much, but certainly not all, mining, oil and gas, manufacturing and service industries), rather than extensive spatial footprint impacts and lower economic profit margins (e.g. much, but not all, agriculture).

In many cases, biodiversity offset approaches could improve a status quo of limited or inadequate compensation for development, so there is a need for a balance between certainty and simplicity in offsetting.

We conclude that in many cases even low-quality, incomplete, impermanent, poorly implemented biodiversity offset approaches could provide more positive outcomes for biodiversity than a status quo (see Introduction) of limited or inadequate compensation. There is often a trade-off between certainty and simplicity: basing an offsetting system on the best available science may lead to it being difficult or impossible to implement because of the transaction costs such a system would incur. Coggan et al. (2013a, 2013b) discuss factors influencing offset transaction costs. Poor implementation has been the main issue identified in the majority of reviews to date (Section 5.3.2), owing to both theoretical and practical constraints to offset opportunity and feasibility (Pilgrim et al., 2013). This will inevitably mean that most development impacts on biodiversity are not fully compensated under offset systems.

ICMM & IUCN (2012) concluded that “Business remains hesitant to invest in offsets due to uncertainty of the outcome as a risk-management tool” – i.e. a perceived lack of scientific/stakeholder consensus on offsets suggests to companies that voluntary offset efforts will not receive broad stakeholder acclaim. This is an unfortunate perception, owing to the inevitability that there will be never be a well-defined one-size-fits-all system and that, in many cases, offset approaches bring added value to biodiversity. We thus believe, on the basis of this review, that – rather than simply critiquing offsets –

the conservation community should more often applaud voluntary offset efforts, actively support attempts to achieve no net loss through best-practice offsets, and provide practical guidance and constructive criticism within a safe learning environment.

In some cases, offsetting may not be the best approach.

Offsetting may not be suitable in cases where existing biodiversity protection, stakeholder involvement in development decisions, and/or adherence to the mitigation hierarchy are already strong and effective. Offsetting may also be problematic in cases where offset policies are largely symbolic yet neutralize environmental concerns (owing to a perception that impacts were being adequately compensated for by offsets), perhaps even weaken pre-existing legislation or practice, or are too complex to implement (for practical or ecological reasons). In such cases, offsetting may degrade current legislation or practice, and facilitate more development consent than may otherwise have taken place (Walker et al., 2009; Gardner et al., 2013). We believe that the conservation community should enter dialogue with governments to ensure that either such inappropriate offset policies are not put in place or that pragmatic best-practice policies are established and well implemented, monitored and enforced.

Current offsetting outcomes would be most improved through integration of societal biodiversity conservation goals, more practical design, greater adherence to the mitigation hierarchy and better implementation.

In regulatory offsetting systems, which comprise the majority of offset implementation experience to date (Madsen et al., 2010), the overriding reason for underachievement of offset outcomes is not bad design, but ineffective implementation. Offsets have failed to achieve their goals in a large proportion of cases. This appears to be due in part to overly theoretical offset designs that lack feasibility testing and stakeholder engagement. Further, there is a lack of incentives for effective implementation, owing to limited capacity in authorities for monitoring, oversight and enforcement – a capacity gap which is reinforced by disincentives for strong regulation (Section 5.3.2).

Two offset design factors would, however, significantly improve current offsetting outcomes. First, offsetting targets and no net loss definitions should be linked to biodiversity conservation goals (and preferably development goals) in separate policies and plans (such as national or provincial systematic conservation plans). If such goals have not yet been well defined, offset programmes and project-level voluntary offsets will need to define such goals from first principles, which is inevitably a suboptimal approach. Second, greater adherence to the rest of the mitigation hierarchy before jumping to use of offsets will – in almost all cases – reduce the use of inappropriate offsets.

A few conditions are necessary for biodiversity offset approaches to provide positive outcomes for biodiversity, irrespective of the concept of no net loss.

Biodiversity offset approaches are generally likely to provide more positive outcomes for biodiversity than a status quo of limited or inadequate compensation, in two main ways. First, offsets would provide better compensation for impacts than limited or inadequate compensation. Second, offsetting would provide an opportunity for better management of biodiversity of conservation concern. This is because: (i) existing land use in areas proposed for offsets may be causing ongoing loss of biodiversity values (e.g. through unsuitable grazing levels); and (ii) existing use rights or insufficient capacity for conservation mean that such management will continue if no offset takes place (Norton, 2007). The following conditions are thus likely to be necessary to ensure offsetting is an improvement over a status quo of limited or inadequate compensation:

- avoidance of in lieu fees for peripheral activities (such payments are commonly given by developers already);
- some form of offset additionality requirements;
- greater regulatory clarity (in jurisdictions with regulations), proper enforcement of regulations (to ensure effective implementation) and learning lessons from implementation; and
- poorly developed biodiversity protection and limited stakeholder engagement in the development process.

Numerous conditions are necessary to achieve no net loss through the implementation of biodiversity offsets.

'No net loss' is a goal in which residual impacts on biodiversity after mitigation do not exceed the gains from offsets. No net loss has no universal definition: it can have varying definitions dependent on what biodiversity and human preferences are accounted for, and how they are accounted for. For example: (i) it may be defined at various spatial scales; (ii) it may select various components of biodiversity; (iii) it may be defined in relation to formal targets or goals for biodiversity conservation in a given country or area; (iv) losses and gains may be compared in a strictly 'like-for-like' way or may allow 'like-for-like or better', meaning inclusion of 'trading up' in terms of conservation concern (e.g. replacing a hectare of house mouse habitat with a hectare of panda habitat). The fact that 'no net loss' goals vary is an important one as different goals necessarily require different conditions.

From this report, we conclude that 20 main conditions are likely to be necessary to ensure offsets achieve their goal of no net loss:

Facilitating conditions (i.e. existing policies and plans):

1. existing definition of societal biodiversity conservation goals (and preferably development goals) in policies and plans (e.g. national or provincial systematic conservation plans). Without pre-existing conservation goals, offset programmes and project-level voluntary offsets would best define such goals from first principles through a stakeholder consultation process, using available guidance (Section 5.1.1).
2. usually, adherence to the mitigation hierarchy: it will always be important to provide guidance on how – and how far – the mitigation hierarchy should be followed (Section 5.1.2).

Scope and spatial scale:

3. inclusion of all biodiversity in a particular 'no net loss' definition within the scope of offsetting (Section 5.2.1).
4. a match between the spatial scale of permitted offsetting and the spatial scale of a particular 'no net loss' definition (Section 5.2.2).

Metrics, currencies, and the ways in which they are exchanged:

5. currencies that include (or represent) all biodiversity within a particular 'no net loss' definition, likely requiring a balance between simple/few and complex/numerous metrics, and multiple (species- and habitat-based) currencies (Section 5.2.3).
6. 'like-for-like' or 'like-for-like or better' exchanges (i.e. not trading down: Section 5.2.4).
7. disaggregation of, or minimum thresholds for, any metrics that are essential to a particular 'no net loss' definition[1] (Section 5.2.4).
8. limits on declines in relative condition/quality[2] will be relevant on a case-by-case basis (Section 5.2.4).

Limits to what can or should be offset:

9. definition of upper limits to what can or should be offset – almost certainly by reference to systematic conservation plans based on clear conservation goals with specific targets (Section 5.2.5).
10. if they are defined, lower limits to what impacts can or should be offset must not exclude impacts/biodiversity which are included within the particular definition of 'no net loss' being used in a given situation (Section 5.2.6).
11. guidance on relative offsetability (i.e. how appropriate or feasible offsets are between lower and upper limits) is not a pre-requisite for achievement of no net loss, but would be a significant factor in likelihood of success of no net loss programmes (Section 5.2.7).

1 In order to avoid critical losses to one essential biodiversity element (e.g. canopy cover) at the expense of another essential element (e.g. standing wood density) owing to combination of all metrics into a single index figure (e.g. '42'). Disaggregation would mean that each disaggregated element would be considered separately. Minimum thresholds would ensure that losses were still within 'safe limits' (not risking extinction).
2 For example, a large medium-condition site is not necessarily a fair offset for loss of a small high-condition site, but such an offset could be suggested by use of aggregated metrics such as habitat hectares.

Additionality:

12. requirement for demonstration of clear additionality, plus methods such as credit adjustments (reductions) to cope with leakage where necessary (Section 5.2.8).

Time considerations: assuring 'permanence' and managing time lags':

13. mechanisms to assure permanence in case of disasters (e.g. insurance, bonds), for long-term management (e.g. long-term financing mechanisms), and to secure land management rights (e.g. easements, convenants, protected area designation, retirement of sub-surface mineral rights) (Section 5.2.9).
14. where no net loss definitions include human time preference, avoidance or reduction of temporal loss through up-front habitat-/species-banking (or, for impacts of relatively low significance on secure biodiversity, multipliers). Otherwise, savings bank approaches are not a pre-requisite for, but would be a significant factor in increasing the likelihood of achieving, no net loss, in situations where they are practical (Section 5.2.10).

Managing uncertainty and risk:

15. comprehensive identification of sources of uncertainty and risk, estimation of the scale of uncertainties, and estimation of the probability and consequences of risks (Section 5.2.12).
16. use of multipliers, insurance/bonds or production of offset gains before impacts to account for lack of precision (Sections 5.2.11, 5.2.12).
17. use of bet-hedging, insurance/bonds or production of offset gains before impacts to account for uncertainty over offset success (Section 5.2.12).
18. use of insurance/bonds to account for uncertainty over whether offset gains can be sustained (Sections 5.2.9, 5.2.12).

Implementation:

19. sufficient capacity to review, implement, and monitor offsets, and to enforce regulations (Section 5.3.2).
20. stakeholder engagement during identification of scope, scale and location of offsetting, and in development of exchange rules (Section 5.3.4).

Some additional conditions will promote optimal efficiency or optimal biodiversity outcomes from offsetting.

A strong no net loss goal could be expected to drive positive biodiversity outcomes. We have identified some additional conditions for optimal efficiency or optimal biodiversity outcomes from offsetting, over and above achievement of no net loss:

- when no net loss is tightly defined, selection of indicators of biodiversity features of conservation concern or direct inclusion of such biodiversity features where no suitable indicators exist;
- offset location determined more by conservation planning than up-front restrictions;
- weighting of biodiversity features in exchange mechanisms, rather than in currencies;
- lower limits to what impacts can or should be offset, below which a simpler compensation system applies;
- up-front provision of offset gains, e.g. through habitat-/species- savings banks; and
- use of multipliers to take a precautionary approach to lack of precision and to achieve conservation goals (and in some cases to address time preference).

3 Introduction

Purpose of this paper

This report was developed as an input paper to inform the IUCN Technical Study Group on Biodiversity Offsets, which is part of a process to develop an IUCN biodiversity offsets policy following Resolution 110. It is intended as a basic introduction to technical issues related to offsets. As such, it points the reader to relevant references in which to find more detailed discussions of the complex issues summarized here. This report is intended as a companion piece to a paper that covers policy options for governments (ten Kate & Crowe, 2014), and thus discussions of policy issues are kept to a minimum in this document.

The paper is a summary of research undertaken for a technical event on biodiversity offsets at the IUCN World Conservation Congress in Jeju, Korea, in 2012. As such, it is a detailed technical – rather than general level – resource paper and addresses points raised in Annex 8 to Council decision C/78/8/b (Section 4). Further, it is not fully up to date with research on biodiversity offsets, but instead synthesizes knowledge until late 2012 (including papers in press at that time, since published in 2013). Some more recent references have been added during subsequent peer review. This paper sets out the opinions and experience of the two authors and is not a reflection of the position of IUCN or its Members.

Background

'Biodiversity offsets' are here considered to be measurable conservation outcomes resulting from actions designed to compensate for significant residual adverse biodiversity impacts arising from project development after other appropriate prevention and mitigation measures have been taken (following BBOP, 2012a). Biodiversity offsets are the last step of the mitigation hierarchy. The goal of biodiversity offsets is to achieve no net loss (or net gain) in biodiversity (e.g. ICMM, 2005; BBOP, 2012a). We note that biodiversity offset definitions vary, and can include compensation for *all*, not just significant, residual impacts (to aim for stricter no net loss goals). Since outcomes are difficult to predict with certainty, and most offsets have not yet fully achieved intended outcomes, we refer to all efforts intended to achieve no net loss as 'biodiversity offsets'. Efforts aimed at only partial compensation of impacts are variously referred to as 'partial biodiversity offsets' or 'compensation' (the latter is also used to refer to offsets in some government policies/legislation).

'No net loss' is a goal in which residual impacts on biodiversity (after other mitigation measures have been taken) do not exceed the gains from offsets. Such a goal may be defined at various spatial scales (e.g. province, country, etc.) and may encompass varying definitions of biodiversity (e.g. all threatened species, all known species, all ecosystem types, etc.) It could be defined in relation to formal targets or goals for biodiversity conservation in a given country or area (Brownlie & Botha, 2009). Losses and gains may be compared in a strictly 'like-for-like' way (e.g. replacing a hectare of house mouse habitat with a hectare of house mouse habitat) or may allow 'like-for-like or better', meaning inclusion of 'trading up' in terms of conservation concern (e.g. replacing a hectare of house mouse habitat with a hectare of panda habitat) – though clearly no biodiversity trade is 100% like-for-like on all levels (e.g. if considering genetic diversity). Pragmatic best-practice is that offsets to achieve no net loss should be 'comparable, additional and permanent' (Gardner et al., 2013). Precautionary approaches to no net loss may include multiple conditions, for example that offsets should have limits and fully address risk and uncertainty (BBOP, 2012a; Gardner et al., 2013). This report considers these conditions and in which cases – and to what extent – they are necessary.

Industry bodies have suggested that a number of areas of uncertainty are hindering uptake of non-regulatory biodiversity offsets (e.g. ICMM 2005a, 2005b). In the last five years, many of these areas of uncertainty have been substantially resolved through concerted research by the Business and Biodiversity Offsets Programme (BBOP), including development of the first best-practice standard for voluntary offsets (BBOP, 2012a), and by increasing implementation of both voluntary and regulatory offsets around the world (Madsen et al., 2010, 2011).

This report assesses lessons learned regarding the conditions required for optimising the contribution of offsets to positive outcomes for biodiversity. It takes as its basis a status quo in which compensation for residual losses of development projects is usually absent or inadequate (Brownlie et al., 2012; Hill & Arnold, 2012). While certainly not the case in every country, this status quo has been the experience of the authors and many of their colleagues, and we feel it is reflected more broadly in the inability of most countries globally to even significantly reduce the rate of biodiversity loss (Butchart et al., 2010). In particular, this report highlights areas of consensus and areas where further research is still needed, by asking the following two questions:

(i) Under what conditions do biodiversity offset approaches provide positive outcomes for biodiversity, irrespective of the concept of no net loss?

Here we identify the conditions that facilitate better outcomes for biodiversity from some form of scientifically-based approach to compensating for impacts, compared with the status quo in which compensation for residual losses of development projects is usually absent or inadequate. These conditions are clearly much broader than those required for offsets to lead to no net loss. Gordon et al. (2011) and Bull et al. (2014) demonstrate the importance of the choice of baseline scenario (i.e. status quo) in evaluating the success of different offset scenarios. Here we define that baseline scenario on the basis of reality globally, wherein development proceeds at the expense of biodiversity (exacerbating a background rate of biodiversity decline): most countries do not have requirements for offsets and so any biodiversity-based compensation that does actually happen as a result of development is often unscientific and poorly quantified. In order to be widely relevant, answers to this question are not always applicable: in a few cases, notably in the USA and Australia, the baseline scenario is actually quite a developed offset system.

(ii) Under what conditions is it possible to achieve no net loss through the implementation of biodiversity offsets?

As explained above, the goal of biodiversity offsets is to achieve at least no net loss of biodiversity. This is a much more ambitious goal than addressed in the first question, and thus requires a much more narrow and strict set of conditions – both technically and politically. Owing to a range of definitions of no net loss, we follow pragmatic best-practice in considering offsets to achieve no net loss as needing to be ‘comparable, additional and permanent’ (Gardner et al., 2013).

In answering these two questions, this report reviews a number of issues that may be required for best biodiversity outcomes to be achieved from offsetting. These are classed as facilitating conditions, technical conditions and implementation conditions. For each issue, the report:

- reviews existing scientific literature and published implementation experience;
- summarizes potential approaches to the issue;
- discusses conditions that are required to improve upon the status quo or achieve no net loss; and
- summarizes these conditions in tabular form.

4 Relationship to Annex 8 to Council decision C/78/8/b

This resource paper addresses all points in Annex 8, but in an edited structure that logically leads through the following sections:

5.1. Facilitating conditions;
5.2. Technical conditions; and
5.3. Implementation conditions.

The relationship between individual bullet points in Annex 8 and sections of this report is mapped in the following table.

Annex 8	Relevant sections of this resource paper
The conservation imperative of the “mitigation hierarchy” (beyond its application to private sector actions	5.1.2
The role of biodiversity offset in the mitigation hierarchy	5.1.2
Demonstrating and verifying additionality beyond business as usual, including in Protected Areas	5.2.8
Accounting for time lags between the biodiversity losses and gains	5.2.10
Addressing the permanence of offset management	5.2.9
Approaches to address the variability in data availability and reliability	5.2.12
Methods, metrics and currencies for quantifying losses and gains of biodiversity at the site level, where appropriate referring to knowledge products mobilized through IUCN, in order to inform exchanges between impact and offset sites	5.2.3
Minimum requirements for assessment of residual losses (e.g. secondary impacts)	5.2.12
Assessment of elements of biodiversity (genetic diversity, species, ecosystem services) to be included in offset calculations: use of irreplaceability and vulnerability as parameters	5.2.1; 5.2.6
Requirements for stakeholder and rights-holder inclusion and consent in biodiversity offset design, development and implementation	5.3.4
Regulatory frameworks and financing mechanisms to guide and fund the implementation of biodiversity offsets	5.2.9; 5.3.1; 5.3.2; 5.3.3
Policy conditions required for optimising the contribution of offsets including the role of landscape or bioregional level planning, biodiversity targets, caps, minimum thresholds, safeguarding threatened species etc.	5.1.1; 5.2.2; 5.2.4; 5.2.5; 5.2.6; 5.2.7; 5.2.11

5 Conditions for offset success

This section covers facilitating (5.1), technical (5.2) and implementation (5.3) conditions. By far the bulk of scientific research has, for obvious reasons, addressed technical conditions. Section 5.2 is thus the most detailed and lengthy part of this report. Facilitating and implementation conditions are not seen as any less important – indeed they may be more important – but are dealt with in greater depth by a companion policy input paper (ten Kate & Crowe, 2014). Here, these issues are addressed more broadly and less conclusively, and are based on the authors' experience to a far greater degree.

5.1 Facilitating conditions

5.1.1 Extent to which societal biodiversity conservation goals and societal development goals are defined in policies and plans

Definition of societal biodiversity conservation goals (and preferably societal development goals) in policies or plans (or some other process led by government and/or receiving broad stakeholder input: Section 5.3.4) is a crucial element of national, provincial or state offsetting programmes, and useful even for informing project-level offsets. Where they exist, these goals de facto: (i) define the offsetting scope (what biodiversity is considered important by society: Section 5.2.1) and spatial scale (within what geographic area is offsetting acceptable to society: Section 5.2.2); (ii) inform rules of exchange (e.g. what biodiversity features are more important than others: Section 5.2.4); (iii) identify limits to what impacts can or should be offset (what are the maximum amounts of biodiversity loss acceptable to society: Section 5.2.5); (iv) ensure consideration of threats to biodiversity (including to offsets); (v) enable prediction of cumulative impacts; and (vi) help to target location of offsets within a landscape (Kiesecker et al., 2009, 2010; McKenney & Kiesecker, 2010; Gordon et al., 2011; Underwood, 2011; Pilgrim et al., 2013a; Saenz et al., 2013). Without existing clarity on societal biodiversity conservation goals, an offsetting programme itself will best identify its scope, scale, limits and rules of exchange through stakeholder consultation – i.e. through identifying societal biodiversity conservation goals. Goals can be – and often are – adopted from existing global frameworks (e.g. BBOP, 2012a; IFC, 2012), but these do not always represent national or local societal goals. Without clarity on such development goals, offset plans may be compromised by ongoing development outside of the scope of the offset system (Saenz et al., 2013). This paper's companion (ten Kate & Crowe, 2014) further discusses integration of mitigation measures and planning processes.

At a basic level, things such as threatened species lists or maps of sites of significance for their contributions to the global persistence of biodiversity (e.g. Key Biodiversity Areas: ICMM & IUCN, 2012) can be considered to represent partial biodiversity conservation goals, if we assume a general societal preference for avoiding irreversible loss of biodiversity (such as extinction of species) – i.e. there should be a corresponding preference for action to address threats affecting threatened species or sites of significance to biodiversity persistence. Progress towards (through offsets) – or away from (through development impacts) – societal biodiversity conservation goals can be measured (Dymond et al., 2008). The utility of such goals is, however, optimized if they are more comprehensive (across biodiversity) and broad-scale (across landscapes). This latter point is important – project-by-project achievement of no net loss will be less effective than planned achievement of no net loss over an entire landscape or country (Bull et al., 2014). Likewise, goals need to be dynamic – adapting to the changing state of conservation and development over time. Many national, state or provincial policies and plans – such as National Biodiversity Strategy and Action Plans – are static. Systematic conservation plans may thus be the optimal place from which biodiversity conservation goals are drawn.

Systematic conservation planning is a stakeholder-driven process for dynamic landscape-level spatial prioritization of sites for biodiversity conservation. Such a system provides an extremely useful basis for land-use planning, in which certain zones are seen as preferable for conservation and offsetting and other zones as preferable for development activities that may remove remnant or low conservation significance habitat (Brownlie & Botha, 2009; Kiesecker et al., 2010; Wissel & Wätzold, 2010; Clare et al., 2011; Obermeyer et al., 2011). Venter et al. (2012) used systematic conservation planning techniques to maximize conservation and development tradeoffs related to REDD+. Nonetheless, Gibbons et al., (2009) point out several operational issues, principally focused on the fact that detailed biodiversity data measured at the site level (to compare impact and offset equivalence) are generally not available at broader spatial scales suitable for systematic conservation planning. This is likely to remain the case in most places, so systematic conservation plans should not be seen as the full solution, but as providing broad zones of suitability/non-suitability for offsets.

In most cases, societal biodiversity conservation goals within systematic conservation plans (or other policies and plans) outline what biodiversity society would like to see conserved, regardless of budget and in the long-term. In some cases, however, goals may be specifically defined for particular budgets – e.g. a realistic five-year protected area establishment plan in a particular country. In such cases, offsetting that simply implemented activities within such a pre-existing plan would likely not be additional (Section 5.2.8).

Conditions required to improve upon the status quo	Properly implemented offsets will often improve upon a status quo of limited or inadequate compensation regardless of the extent to which biodiversity conservation goals and societal development goals are defined in policies and plans.
Conditions required for no net loss	No net loss would most easily be achieved when biodiversity conservation goals and societal development goals are already defined in policies and plans. Without pre-existing goals, offset programmes and project-level voluntary offsets will best define such goals from first principles through a stakeholder-driven process, using available guidance.

5.1.2 Extent to which a process for mitigation and offsets is defined

The mitigation hierarchy is the logical, sequential framework in which impacts are first avoided, then minimized, remediated, and finally any residual impacts are offset. It was incorporated twenty years ago into the Convention on Biological Diversity (1992) but has been active policy in Germany and the USA since the 1970s. Application of the mitigation hierarchy is fundamental to environmental best-practice (e.g. IAIA, 2005; McKenney & Kiesecker, 2010; BBOP, 2012a). This paper's companion (ten Kate & Crowe, 2014) further discusses integration of mitigation measures and planning processes.

It is generally agreed that biodiversity offsets should only be used as the final step of a mitigation hierarchy (Norton, 2009; Bekessy et al., 2010; McKenney & Kiesecker, 2010; Quétier & Lavorel, 2011; BBOP, 2012a). While this principle is often enshrined in legislation (McKenney & Kiesecker, 2010), and the high cost of effective offsets should actually provide economic incentives for reducing offset liabilities through prior mitigation (Houdet et al., 2012), the extent to which a mitigation hierarchy is actually used has varied, and there have been numerous cases in which offsets have been preferred over avoidance on social or economic grounds (Hough & Robertson, 2009; Walker et al., 2009; Clare et al., 2011). Moreover, there is a lack of clarity on precisely where this 'final step' is (Walker et al., 2009) – i.e., when the other mitigation steps are considered 'complete' and offsets are permitted to compensate for residual impacts. BBOP (2012a) state that offsetting is "a measure of last resort" to be deployed "after appropriate avoidance, minimization and on-site rehabilitation measures have been taken according to the mitigation hierarchy", while Treweek (2009) suggests offsets should be deployed after "using all reasonable and cost-effective prevention and mitigation measures". But how should 'appropriate', 'reasonable' and 'cost-effective' be defined? In other words, how far should avoidance and minimization be taken when they trade off against a development project's plans? (Theoretically they could be taken to the point of the project no longer existing.) Under offset systems that allow 'trading up' (Section 5.2.4), it may not always even be most beneficial for conservation to fully pursue other mitigation steps – for example, of impacts on biodiversity of low conservation concern if such impacts can instead be compensated for by conservation of biodiversity of high conservation concern (posing methodological challenges: Section 5.2.4). In the absence of guidance from a competent authority, planning authorities and other decision-makers are left to make value judgements on a case-by-case basis.

The principle of choosing the most cost-effective mitigation (including offset) measures is simple (Dickie &Tucker, 2010: pp. 92-93). In only a few cases, however, will enough data exist to quantitatively compare effectiveness of mitigation (including offset) measures: Igual et al. (2009) present one example, using real-life data to assess proposals for offsetting marine seabird bycatch through rat eradication on nesting islands, and show that in these cases such measures will ultimately fail owing to insufficient application of the mitigation hierarchy (i.e. reducing bycatch in the first place). In most cases, the trade-off between other mitigation steps and offsets is less clear. Kiesecker et al. (2010) and Obermeyer et al. (2011) start to address this question on a landscape scale (i.e. emulating the role of a government planning department) by using conservation planning techniques and modifying target portfolios to avoid areas of high development potential where possible. Where not possible, i.e. where sites in areas of high development potential are key to meeting conservation goals, avoidance and minimization by development projects are seen as priorities. Pouzols et al. (2012), though omitting a spatial dimension for simplicity, take such approaches further by considering optimal allocation of resources across habitat maintenance, conservation management, restoration, and biodiversity offsetting.

Concerns have been raised that offsets may be seen as a ‘license to trash’ by offering a way to sidestep implementation of earlier steps in the mitigation hierarchy (McKenney & Kiesecker, 2010). Avoidance and minimization, the first steps in the mitigation hierarchy, will usually be optimal for biodiversity – sidestepping avoidance and mitigation guarantees biodiversity loss yet can only offer uncertain promises of gains from restoration or offsets.

Conditions required to improve upon the status quo	Properly implemented offsets will often improve upon a status quo of limited or inadequate compensation regardless of the extent to which a process for the mitigation hierarchy (including offsets) is defined (although any definition, or application, of the mitigation hierarchy would usually further improve upon the status quo).
Conditions required for no net loss	To facilitate achievement of no net loss, it will always be important to provide guidance on how – and how far – the mitigation hierarchy should be followed. In some cases, later steps in the hierarchy may be more appropriate or feasible in order to optimize conservation outcomes.

5.2 Technical issues

Throughout Section 5.2, it should be borne in mind that the complexity of technical restrictions and exchange rules in an offsetting system will be inversely proportional to the amount of trade (Salzman & Ruhl, 2000). Development of ecologically-defensible offset design, that can reasonably be expected to achieve no net loss, may result in a system that is too costly or difficult to implement.

5.2.1 Biodiversity scope

Generally accepted definitions of biodiversity include variability in composition, structure and function (e.g. Noss, 2001) of genes, species and ecosystems, and the interactions between them (see for example, the Convention on Biological Diversity[3]). Practically, however, it would be impossible for offset programmes to ask for consideration of every single component of biodiversity, particularly if they aim for like-for-like offsetting (Section 5.2.4). Instead, programmes typically focus on more narrow definitions of biodiversity, which may be restricted by factors such as type (e.g., habitats/ecosystems, species or processes/functions), taxonomy (e.g., what combinations of vertebrates, invertebrates, fungi and plants are included), scale (e.g., fine- or broad-scale ecosystem classifications, species, subspecies or populations), conservation status (e.g., only threatened species/ecosystems), or value type (e.g., only existence values of biodiversity or also ecosystem services) (Burgin, 2008; Quétier & Lavorel, 2011). Such narrower definitions are most often chosen on the basis of highest conservation concern and/or on the basis that selected biodiversity components are representative of biodiversity as a whole. A practical example can be found in the Critical Habitat Assessment developed as part of the Oyu Tolgoi Environmental and Social Impact Assessment (TBC & FFI, 2012).

3 www.cbd.int

Assessments of conservation concern are most frequently focused around the concepts of irreplaceability and vulnerability (Margules & Pressey, 2000; Wilson et al., 2005; Brooks et al., 2006). Knowledge products mobilized through IUCN[4] are of great relevance in identifying biodiversity components of highest conservation concern, e.g. via the Red Lists (of threatened species and ecosystems: e.g. Rodrigues et al., 2006; Rodriguez et al., 2011), Protected Planet (including the World Database on Protected Areas: e.g. Butchart et al., 2012) and the Standard for identification of areas of global significance for biodiversity (such as Key Biodiversity Areas: e.g. Eken et al., 2004; Butchart et al., 2012). Conversely, it is likely to be difficult to find one-size-fits-all indicators of biodiversity as a whole or of biodiversity of conservation concern – for example, Matthews & Endress (2008) point out that a US wetland restoration indicator of ‘percentage of hydrophytic plant cover’ has incentivized creation of deeper, wetter wetlands than those originally impacted. Case-by-case selection of biodiversity components or indicators thought to be representative is thus likely to be the most useful approach, but its value is tempered by the impracticality of such an approach in biodiversity banking (which requires transferable indicators).

Poorly-known biodiversity (such as fungi and invertebrates) and biodiversity that is particularly difficult to measure (such as genetic diversity, or process and function) are most likely to be excluded on practical grounds (although approaches to tackle these have been proposed, e.g. Landscape Equivalency Analysis for gene flow: Bruggeman et al., 2009). Ecosystem services are also often excluded from offset considerations, or given only passing consideration, because they are less likely to be replaceable at different locations (Palmer & Filoso, 2009) and are sometimes substitutable (e.g. mains water provision may substitute for some functions of a wetland) and thus like-for-like exchanges may not be preferred – although substitution is less likely to be effective for economically poorer human communities and for cultural ecosystem services (Millennium Ecosystem Assessment, 2005). Indeed, substitution (essentially an analogue of trading up; Section 5.2.4) of ecosystem service values can conflict with compensation of intrinsic biodiversity values (Levrel et al., 2012). Only limited research to date has addressed the extent to which ecosystem service values are substitutable for humans (e.g. Sherren et al., 2012).

Summary of approaches to defining biodiversity scope:

- limited but simple definitions of biodiversity; and
- inclusive but complex definitions of biodiversity.

Conditions required to improve upon the status quo or achieve no net loss:

Given a status quo of limited or inadequate compensation for development project impacts, any offsetting will likely be an improvement – even if some biodiversity is excluded by narrow definitions of biodiversity, or if offsetting programmes are unnecessarily complex owing to broad definitions of biodiversity. To achieve no net loss, the scope of biodiversity features to be mitigated (including offset) has to include (or represent via good surrogates) all those features within the relevant definition of ‘no net loss’ (Quétier & Lavorel, 2011). An offset system would, however, be most beneficial if it is inclusive enough to represent all biodiversity features of conservation concern yet without inclusion of too many additional features that add more complexity than they do value. This is a fine balancing act that is likely to be best solved by selection of surrogates of biodiversity features of conservation concern (for example, habitat is likely to be a reasonable surrogate for many species) and direct inclusion of such biodiversity features where no suitable surrogates exist. Practically, biodiversity that is poorly known or very difficult to measure, as well as ecosystem services, may best be excluded from offsetting systems in order to ensure they function effectively.

Conditions required to improve upon the status quo	None related to the scope of biodiversity included in offsetting – properly implemented offsets will often improve upon the status quo.
Conditions required for no net loss	The scope of biodiversity must include (or represent) all biodiversity within a particular definition of ‘no net loss’.
Additional notes	When no net loss is tightly defined, optimal offsetting would select indicators of biodiversity features of conservation concern or directly include such biodiversity features where no suitable indicators exist.

4 http://iucn.org/about/work/programmes/ecosystem_management/ipbes/ipbes_and_iucn/knowledge

5.2.2 Spatial scale

'Spatial scale', or the geographic area within which offsetting is acceptable in relation to the area of impacts, should be distinguished from 'location' of offsets – which defines optimal offset locations through characteristics such as connectivity (e.g. Quétier & Lavorel, 2011). Spatial scale is a component of most exchange mechanisms (Section 5.2.4), but is such an up-front consideration in most offsetting (e.g. will international offsets be allowed?) that it is dealt with separately first here.

It is often recommended that offsets be established close to impact areas in order to increase the likelihood of approximating to like-for-like exchanges of ecological communities (because of turnover of biodiversity across landscapes) and to increase the likelihood of compensating the same biodiversity beneficiaries (e.g. people benefiting from local ecosystem services or local biodiversity existence values) as are impacted (Brownlie & Botha, 2009; Kiesecker et al., 2009; Salzman & Ruhl, 2000; McKenney & Kiesecker, 2010; BenDor & Stewart, 2011; Womble & Doyle, 2012). Further, political issues ensure that offsetting rarely occurs across disparate jurisdictions – especially across national boundaries (Edwards & Laurance, 2012). 'Functional areas' or 'service areas' for offsetting, where they exist (particularly in the USA), are often based primarily on ecological factors such as the genetic distribution of a species, watershed configuration or soil attributes (Hill, 2008; Womble & Doyle, 2012; Pilgrim et al., 2013a). Nonetheless, from a conservation point of view, optimal offset sites (in either like-for-like or trading up scenarios; Section 5.2.4) will not always be near impact sites. Choice of offset location can thus be informed by conservation planning exercises rather than constrained by *a priori* spatial scale definition (Brownlie & Botha, 2009; Kiesecker et al., 2009, 2010; McKenney & Kiesecker, 2010). However, the extent to which this information helps may be limited (Gordon et al., 2011), transaction costs are increased, and it is important to ensure a transparent process appropriate to the capacity of stakeholders (Section 5.3.2). In addition, Womble & Doyle (2012) note that unbundling of impacts (into various species, habitats, ecosystem services, etc.) would facilitate identification of multiple suitable offset locations, and thus promote offsetting as a viable market-based mechanism.

Another issue of spatial scale is the scale of the goal of offset systems: Bull et al. (2014) discuss the differences between expecting no net loss project-by-project and across an entire landscape/ jurisdiction. Such issues should be clearly outlined in goals (Section 5.1.1).

Summary of approaches to defining spatial scale:

- only broad geo-political limits.
- definition of 'functional areas' or 'service areas'.
- few up-front spatial scale limits – i.e. little or no geographic restriction – optimal offset location instead determined by conservation planning exercises.

Conditions required to improve upon the status quo or achieve no net loss:

Given a status quo of limited or inadequate compensation for development project impacts, any offsetting will be an improvement even if only broad geo-political limits are given to the spatial scale of offsetting. To achieve no net loss, the spatial scale of permitted offsetting must match the spatial scale of the definition of 'no net loss' in a particular situation – this might simply be a broad geo-political unit such as a state or country. An offset system would, however, be most beneficial if optimal offset location was determined more by conservation planning than up-front definitions of permitted spatial scale of offsetting. Nonetheless, some up-front restriction of offsetting to within geo-political units – and, for biodiversity valued by humans, to within functional/service areas – will usually be desirable.

Conditions required to improve upon the status quo	None related to the spatial scale of offsetting permitted – properly implemented offsets will often improve upon the status quo.
Conditions required for no net loss	The spatial scale over which offsetting is permitted must match the spatial scale specified by the relevant 'no net loss' definition.
Additional notes	Up-front restriction of offsetting to within geo-political units – and, for biodiversity valued by humans, to within functional areas – will be necessary, but optimal offset location should be determined more by conservation planning than up-front restrictions.

5.2.3 Measurement of biodiversity (metrics, methods and currencies)

One of the key areas of debate regarding offsets has been the measurement of biodiversity, i.e. the metrics (the parameters used to measure biodiversity, e.g. area, number of individuals, vegetation height, canopy cover), the methods (the ways in which those metrics are actually calculated in the field), and currencies (the units ultimately used in exchanges of losses of, and gains in, biodiversity, e.g. 'habitat hectares'). The terms 'metrics', 'methods' and 'currencies' are not used consistently in offsetting literature and so definitions used here (i.e. as in the previous sentence) may differ from those in other publications. The same metrics and currencies need to be used for assessing loss and gain in any transaction in order to attempt equivalence (Quétier & Lavorel, 2011). In practice, gains may be more rarely weighed against quantified measures of losses than they are qualitatively negotiated by stakeholders (Brown et al., 2014).

Biodiversity is complex and any measure of biodiversity will thus be an imperfect representation of, and an imprecise surrogate for, all variation in all components of a particular area (Salzman & Ruhl, 2000; Walker et al., 2009; Pawliczek & Sullivan, 2011). The key discussion around measurement of biodiversity in offsetting has been the search for 'equivalence' – i.e. defining fungible currencies that facilitate exchange of the same types and amounts of biodiversity in offsets to that impacted. Measuring equivalence is important even in 'trading up' exchanges (Section 5.2.4). In searching for the best practical currencies, Gardner et al. (2013) suggest that they should be "...based on direct, disaggregated and context-dependent measures of biodiversity that provide the most unambiguous and locally-relevant data (e.g. persistence probabilities of a regionally threatened species)." In a similar vein, Dymond et al. (2008) demonstrate that measures of progress towards national conservation goals are technically feasible. There is, however, a need within biodiversity offsetting to balance elaboration of ever more complex metrics (that more accurately and precisely represent biodiversity) with assurance of sufficient simplicity for practical application and exchange (Salzman & Ruhl, 2000; Fennessy et al., 2007; Wissel & Wätzold, 2010; Quétier & Lavorel, 2011). Practical methods are particularly important in the case of habitat-/species-banking approaches, where single big offsets are often used to compensate for multiple separate smaller impacts (Stein et al., 2000). Parkes et al. (2004) also point out the need to develop metrics and currencies in such a way as to "simplify the environment to an extent that a wide range of people can understand the idea". Whatever metrics are used, they need to include appropriate indicators of key variables that are: (i) of interest to both biodiversity and humans (Sherren et al., 2012); (ii) transparent (Norton, 2009); and (iii) standardized for consistent replication (Quétier & Lavorel, 2011). Metrics are either combined (e.g. 'habitat hectares') or used separately to produce 'currencies', so called as they are used for exchange (e.g. BBOP, 2012a) – or, more correctly, owing to their inherent imperfection, for barter (Salzman & Ruhl, 2000; Walker et al., 2009).

In practice, in an attempt to represent habitat complexity in a simple fungible way, multiple biodiversity metrics (e.g. canopy height, canopy cover, understorey height, density of tree holes, density of large trees) are often combined into a single currency (e.g. 'habitat hectares' or 'quality hectares': Parkes et al., 2003; Temple et al., 2012[5]), with at least some metrics likely to be combined in a multiplicative way (Stein et al., 2000; McCarthy et al., 2004; Gibbons & Freudenberger, 2006). However, when multiple attributes are combined into one currency, the single final score can mask critical losses sustained by some elements of biodiversity – i.e. individual metrics or attributes are not substitutable (McCarthy et al., 2004; Gibbons & Lindenmayer, 2007; Bedward et al., 2009; Pawliczek & Sullivan, 2011; Gardner et al., 2013) There is thus a need for exchange rules (Section 5.2.4) that set minimum thresholds for key metrics (Gardner et al., 2013), in order that minimum acceptable limits are set on each key metric, such as density of tree cavities or herbaceous cover, and one cannot be wholly lost owing to high values for others. Alternatively, particularly where metrics apply to different kinds of biodiversity feature (e.g. individual species, ecosystem services or habitats) equivalence would be optimized by keeping key metrics separate (disaggregated) in multiple, complementary currencies (Quétier & Lavorel, 2011; Temple et al., 2012; Gardner et al., 2013)

To address the fact that target species often respond to offset delivery techniques at different rates than target habitats, it is generally recommended that indicators include both structural and biotic considerations. This could be through something as simple as integration of biotic indices into a single indicator (Spieles et al., 2006) through to separate case-specific indicators such as bird lek (display ground) count data (Doherty et al., 2010). Nonetheless, for pragmatic reasons, the majority of currencies are habitat-based (Treweek et al., 2010). Some longer-established offsetting programs are still simplistically based on area alone (Fox & Nino-Murcia, 2005; Quétier & Lavorel, 2011). More common, particularly in recent offsetting, are 'extent × condition' currencies that combine measures of the extent (area) of impacts or offsets with

5 Another practical example can be found in the Net Positive Impact Forecast developed as part of the Oyu Tolgoi Environmental and Social Impact Assessment, in the Biodiversity Appendices: http://www.ot.mn/en/node/2679

measures of the condition (quality) of the same areas (e.g. Parkes et al., 2003; Temple et al., 2012). For example, 100 ha of 25% quality would be viewed as equal to 50 ha of 50% quality. One common concern raised about 'extent × condition' currencies is that they allow replacement of a small area of high quality habitat with a large area of low quality habitat, or replacement of a large area of habitat with a small area of higher quality habitat – these issues are best addressed by exchange rules (Section 5.2.4). Condition may be based entirely on habitat structural characteristics (e.g. Temple et al., 2012), or may include species composition or abundance of indicator species (e.g. Normander et al., 2012).

Increasing evidence for non-linear relationships between biodiversity and area (e.g. He & Hubbell, 2011) casts doubt on the theoretical underpinning of 'extent × condition' currencies. In practice, this means that such currencies are resulting in unequal trades of biodiversity loss and gain. More accurate currencies are necessary to solve this issue, but they will need to balance certainty of outcomes with simplicity of comprehension and application (Parkes et al., 2004). It is quite possible that current practical issues (e.g. with sampling design, statistical power and benchmarks) introduce considerably higher error than such flaws in underlying theory.

Parkes et al. (2003) and McCarthy et al. (2004) discuss issues with using benchmarks of quality/condition (i.e. expressing quality as a percentage of 'original' or 'pristine' quality of a given habitat), particularly in cases where good quality habitat is a consequence of repeated disturbance or where change is the norm. Benchmarks are, however, the only practical way of scoring different vegetation types in a comparative manner (Parkes et al., 2004; Gardner, 2010; Gardner et al., 2013). Quétier & Lavorel (2011) suggest a slightly different approach, in which multiple separate quality benchmark indicators are used and, precautionarily, that which gives the highest offset requirement is used.

Habitat-based currencies can be modified to include information on suitability of different areas/habitats for species of conservation concern (Burrows et al., 2011). In addition, refined indicators of condition/quality such as landscape spatial structure (e.g. fragmentation or genetic variability) can help to capture variance in migration ability or reproductive rate among various impact and potential offset areas (Bruggeman & Jones, 2008; Bruggeman et al., 2005, 2009). Nonetheless, habitat indicators cannot encompass all impacts (e.g. road mortality of a species) and species-specific (e.g. population-based) currencies are often also desirable ('disaggregated currencies' as discussed above). One example is 'Units of Global Distribution' (Temple et al., 2012), area-based measures of species' range which aim to highlight relative importance in a global context. For example, loss or gain of 1 ha of a species' distribution is much more significant if that species is globally restricted than if it is widespread.

Some actual or proposed currencies include not only measures but also weightings, in order that conservation concern (Oliver et al., 2005; Fennessy et al., 2007; Treweek et al., 2010) or other expressions of stakeholder preference (Hajkowicz & Collins, 2009) have a stronger influence on final metrics. A simpler and clearer approach is to incorporate stakeholder preferences into exchange rules (Section 5.2.4): inclusion of yet more information into an already complex single metric seems likely to further confuse stakeholders and mask critical losses by some elements of biodiversity (as discussed above). Knowledge products mobilized through IUCN[6] – such as the Red Lists (of species and ecosystems), Protected Planet (including the World Database on Protected Areas) and Standard for identification of areas of global significance for biodiversity (e.g. Key Biodiversity Areas) – are thus of more relevance to the scope of biodiversity offsets (Section 5.2.1) and the rules by which biodiversity is exchanged (Section 5.2.4).

Financial currencies, i.e. in lieu fees, are used as an alternative to physical offsets in some places (e.g. Wilkinson, 2009; ten Kate & Crowe, 2014). While offering a potentially appropriate solution to offsetting cumulative impacts of low significance (Section 5.2.6), there is significant potential for in lieu fees to never be directed to 'real' offsets (i.e. those producing measurable biodiversity gains) – instead funding peripheral activities such as research or non-additional ongoing government conservation management, or even being appropriated outside of conservation budgets (BenDor & Riggsbee, 2011).

Summary of approaches to measurement of biodiversity:

- simple/few (low equivalence, low transaction cost) or complex/numerous (high equivalence, high transaction cost) metrics;
- single (combined) or multiple (disaggregated) currencies;
- species-based and/or habitat-based metrics and currencies;
- inclusion/exclusion of factors such as measures of conservation concern or stakeholder preference; and

6 http://iucn.org/about/work/programmes/ecosystem_management/ipbes/ipbes_and_iucn/knowledge

- in lieu fees.

Conditions required to improve upon the status quo or achieve no net loss:

Given a status quo of limited or inadequate compensation for development project impacts, almost any offsetting will be an improvement regardless of the design or quality of metrics and currencies. The exception is in lieu fees, which are commonly given (Wilkinson, 2009) yet are often used for peripheral activities that do not provide real measurable biodiversity gains. To achieve no net loss, currencies have to include (or represent via good surrogates) all biodiversity included within the definition of 'no net loss' in a particular situation. Practically, achievement of no net loss is likely to require a balance between simple/few and complex/numerous metrics, as well as multiple disaggregated (both species- and habitat-based) currencies (Gardner et al., 2013). If biodiversity features are to be weighted (i.e. to account for stakeholder preferences such as measures of conservation concern), doing so within exchange mechanisms – rather than within currencies – would best assure a functional offset system.

Conditions required to improve upon the status quo	Properly implemented offsets that are better than in lieu fees for peripheral activities will often improve upon the status quo regardless of the design or quality of metrics and currencies.
Conditions required for no net loss	Currencies must include (or represent) all biodiversity within a particular definition of 'no net loss', likely requiring a balance between simple/few and complex/numerous metrics, and multiple (species- and habitat-based) currencies.
Additional notes	Offsetting would likely be most functional if exchange mechanisms, rather than currencies, address weighting of biodiversity features.

5.2.4 Exchange mechanism (exchange rules)

Currencies (Section 5.2.3) are constructed to facilitate exchange of biodiversity, but cannot incorporate all stakeholder desires and preferences without becoming overly complex. Therefore, there is usually a need for an exchange mechanism – the set of rules by which biodiversity losses and gains are exchanged (or, more correctly, bartered: Salzman & Ruhl, 2000; Walker et al., 2009). Ultimately, a large number of issues can be considered to be relevant to exchange rules, including:

i. 'spatial scale', i.e. the area over which offsetting is acceptable in relation to the area of impacts, dealt with in Section 5.2.2.
ii. limits to what impacts are offsetable, dealt with in Sections 5.2.5-5.2.7.
iii. 'additionality', dealt with in Section 5.2.8.
iv. 'permanence', dealt with in Section 5.2.9.
v. temporal loss, dealt with in Section 5.2.10.
vi. use of multipliers, dealt with in Section 5.2.11.
vii. ways of addressing uncertainty and risk, dealt with in Section 5.2.12.
viii. 'like-for-like' versus 'trading up' exchanges.
ix. substitutability of metrics.
x. limits to declines in quality and area between impact and offset sites.

The first seven of these issues are dealt with in turn in the subsequent sections of this report (encompassing most of the technical issues discussed in Section 5.2). This section focuses on the last three issues (viii-x, above). Issue viii is fundamental to the structure of biodiversity exchanges, while issues ix and x relate more to inherent weaknesses of metrics and currencies (Section 5.2.3).

A preference for, or insistence on, 'like-for-like' or 'in kind' (i.e. comparable or 'equivalent' in the type of biodiversity) exchanges can be found in most offsetting systems. This is understandable, since there is an innate sense of equity in replacing types of biodiversity (whether individual species, habitats, functions, etc.) that are impacted with the same types of biodiversity in an offset. In practice, owing to the complexity of biodiversity and thus inherent imperfection of offset currencies (Section 5.2.3), true like-for-like exchanges are impossible. In particular, it is being increasingly recognized that restoration, even through assisted regeneration, is unlikely to produce like-for-like habitats in all but the most simple ecosystems (Wilkins et al., 2003; Hilderbrand et al., 2005; Gibbons & Lindenmayer, 2007; Palmer & Filoso, 2009; Rey Benayas et al., 2009; Curran et al., 2014). Some 'out of kind' ('trading down' or 'trading up') exchanges may, in any case, be desirable or necessary. For similar reasons of equity, 'trading down' of biodiversity (e.g., exchanging a similar area of common, widespread habitat in an offset for impacts on a rare, restricted habitat) is unlikely to be proposed as a general option in an offsetting system, although it may be permitted by regulators in cases where developments are of high strategic value yet impacts are not truly offsetable (Section 5.2.5). However, 'trading up' (i.e. 'like-for-like or better') of biodiversity is an offset strategy in some countries (Treweek, 2009), and has considerable potential if robust methods can be developed for quantifying exchanges of different biodiversity (Quétier & Lavorel, 2011; Habib et al., 2013). Very few generally-applicable 'trading up' methods have been developed to date, and none have been implemented in practice.

'Trading up' refers to impacting biodiversity of lower conservation concern, while conserving/restoring biodiversity of higher conservation concern (Kiesecker et al., 2010; Wissel & Wätzold, 2010). The term 'trading up' is sometimes also used to refer to impacting biodiversity of lower quality (degraded, smaller patch sizes/populations), while conserving/restoring biodiversity of the same type but of higher quality (more pristine, larger patch sizes/contiguous populations: Dickie & Tucker, 2010; Kiesecker et al., 2010; Wissel & Wätzold, 2010; Edwards & Laurance, 2012), but here we restrict the term to trading up in type (in line with BBOP, 2012a). It has sometimes been proposed that currencies include weightings of conservation concern or other expressions of stakeholder preference (Section 5.2.3), but these are considerations of trading up and are thus best dealt with within exchange rules. Relative conservation concern would best be identified via systematic conservation planning, in which each individual site/biodiversity feature is ranked against every other, or via extinction risk/persistence (Overton et al., 2012). Apart from those ideals, other existing stratifications of conservation concern can play a role (e.g. Dickie & Tucker, 2010: Figure ES.1). Key examples of such stratifications are knowledge products mobilised through IUCN[7] such as the Red Lists (of species and ecosystems: e.g. Rodrigues et al., 2006; Rodriguez et al., 2010), Protected Planet (including the World Database on Protected Areas: e.g. Butchart et al., 2012) and the Standard for identification of areas of global significance for biodiversity (such as Key Biodiversity Areas: e.g. Eken et al., 2004; Butchart et al., 2012).

Once metrics/indicators of relative conservation concern have been identified, a number of potential mechanisms exist for facilitating a process of trading up (e.g. Ludwig & Iannuzzi, 2006; Overton et al., 2013). These methods often present options not only for trading up, but also for the implementation of exchange rules more generally. For example, Overton et al. (2013) develop a means of exchange ('Net Present Biodiversity Value') that provides a basis for determining equity across type, space and time. Nonetheless, such methods have not yet been tested broadly in reality, and would require substantial contextual information about both impacted and offset biodiversity – information that would be both challenging and costly to collect.

While the significance of initial impacts should not be ignored, trading up and/or judicious use of offset multipliers (Section 5.2.11) offers significant novel funding opportunities for biodiversity conservation (Treweek, 2009; Kiesecker et al., 2010; Temple et al., 2010). Moreover, Harper & Quigley (2005) note that insistence on a like-for-like approach in highly-disturbed landscapes is not always advisable because chances of offset success are limited by existing ecological or biophysical bottlenecks. Overall, there is increasing recognition that conservation goals are rarely best served by absolute adherence to a like-for-like approach: if robust methods for exchanging different biodiversity can be developed, trading up may be optimal for impacts on common and widespread biodiversity, while like-for-like may be optimal for biodiversity of conservation concern (McKenney & Kiesecker, 2010; Wissel & Wätzold, 2010). However, in the former case, benefits to rarer biodiversity would have to be weighed against declines of common biodiversity which may be key to ecosystem function yet have limited legislative protection (Pilgrim et al., 2013b; Regnery et al., 2013).

As already mentioned, when multiple attributes are combined into one currency, the single final score can mask critical losses sustained by some elements of biodiversity. In other words, individual metrics or attributes are not substitutable (McCarthy et al., 2004; Gibbons & Lindenmayer, 2007; Bedward et al., 2009; Pawliczek & Sullivan, 2011; Gardner et al.,

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7 http://iucn.org/about/work/programmes/ecosystem_management/ipbes/ipbes_and_iucn/knowledge

2013) Proposed solutions to this (Section 5.2.3) have been to either disaggregate all valued metrics (Quétier & Lavorel, 2011; Temple et al., 2012; Gardner et al., 2013) or set minimum thresholds for individual valued metrics (Gardner et al., 2013).

Limits to declines in quality and area between impact and offset sites have been proposed (Gardner et al., 2013) to ensure that offsetting does not result in either (i) cumulative gains in habitat area, but loss of high quality habitats that may be difficult or impossible to restore, or (ii) gains in habitat quality, but reductions in overall area of habitat. For example, wetland mitigation banking is likely overall leading to gains in wetland area but losses in wetland quality (Kozich & Halvorsen, 2012). Conservation of remaining areas of low quality may sometimes be viewed as crucial to ensuring representation of a given biodiversity type across the landscape (McCarthy et al., 2004) but may more often be viewed as a poor substitute for loss of areas of high quality.

Summary of approaches:

- ‘like-for-like’, ‘trading down’ and ‘trading up’;
- disaggregated valued metrics or minimum thresholds set for valued metrics; and
- limits, or no limits, to declines in quality between impact and offset sites.

Conditions required to improve upon the status quo or achieve no net loss:

Given a status quo of limited or inadequate compensation for development project impacts, any offsetting except trading down will be an improvement regardless of whether there are exchange rules on aggregation of – or minimum thresholds for – metrics, and whether any limits are established for declines in quality between impact and offset sites. To achieve no net loss, either ‘like-for-like’ or ‘like-for-like or better’ are appropriate, depending on the particular definition of ‘no net loss’ (though trading up within the latter requires robust metrics, exchange mechanisms and implementation). For any metrics essential to a particular definition of ‘no net loss’, it will be essential to either disaggregate these metrics or set minimum thresholds. Limits on declines in quality between impact and offset sites are an issue that requires consideration on a case-by-case basis, facilitated by existing principles and guidance.

Conditions required to improve upon the status quo	Properly implemented offsets, except trading down, will often improve upon the status quo regardless of whether there are exchange rules on aggregation of – or minimum thresholds for – metrics, and of whether any limits are required to declines in quality between impact and offset sites.
Conditions required for no net loss	‘Like-for-like’ or ‘like-for-like or better’ (though trading up requires robust metrics, exchange mechanisms and implementation). For any metrics essential to a particular definition of ‘no net loss’, disaggregation or minimum thresholds. Limits on declines in quality on a case-by-case basis.

5.2.5 Upper limits to what impacts can or should be offset

It is generally accepted that there are limits to what can be offset: some residual impacts cannot be fully offset owing to the inherent vulnerability or irreplaceability of the affected biodiversity (Brownlie & Botha, 2009; Gibbons et al., 2009; Norton, 2009; Bekessy et al., 2010; BBOP, 2012a, 2012c). At the extreme, offsets would not be possible for impacts that cause global extinction (BBOP, 2012a, 2012c) or commit biodiversity to extinction by pushing it beyond critical thresholds (Swift & Hannon, 2010), and may not be practical to achieve in other situations (e.g. for habitats where no additional area is available for restoration/conservation, owing to lack of relevant proven offset delivery techniques or inadequate plans/funding; Pilgrim et al., 2013a). There are other cases where they may be considered inappropriate because of the level of risk to biodiversity (e.g. owing to long time lags or high risks of failure; Pilgrim et al., 2013a). Such cases reflect maximum levels of biodiversity loss acceptable to society (Bull et al., 2013).

Upper limits to what impacts can or should be offset can thus be seen as required, to identify cases where impacts are not offsetable because offsets are either theoretically impossible (ecologically non-offsetable), practically unachievable, or socially unacceptable. Impacts (and thus offsets) would ideally not be permitted above such limits, although in practice it is likely that some development benefits (e.g. matters of national security) will always be considered more important than even irreplaceable, irreversible biodiversity impacts. If upper limits are strictly applied, and trading of offset credits is allowed, they effectively operate in a similar way to a 'cap-and-trade' policy (Salzman & Ruhl, 2000; Bruggeman et al., 2005; Maron et al., 2010) and will help to define where avoidance should be prioritised (Clare et al., 2011).

Situations where offsets are theoretically impossible or socially unacceptable are ideally defined by stakeholder-designed systematic conservation plans that prioritise conservation action based on a clear conservation goal with specific targets (Pilgrim et al., 2013a; Section 5.1.1). For example, in Western Cape Province, South Africa, upper limits to what impacts should be offset have been defined in order to ensure that the cumulative impact of development does not cause any ecosystem to become more threatened than 'endangered' and the conservation status of species and 'special habitats' does not decline (Brownlie & Botha, 2009).

Summary of approaches to upper limits to what impacts can or should be offset:

- no upper limits defined;
- upper limits defined by reference to systematic conservation plans that identify conservation priorities based on a clear societal biodiversity goal with specific targets; and
- upper limits defined in another way (e.g. through top-down decisions).

Conditions required to improve upon the status quo or achieve no net loss:

Given a status quo of limited or inadequate compensation for development project impacts, any offsetting will be an improvement – even if some impacts are theoretically or practically not offsetable. Whatever the definition of 'no net loss' in a particular situation, some impacts will always be non-offsetable because they are either theoretically impossible, practically unachievable, or socially unacceptable. Definition of upper limits to identify, and exclude, these non-offsetable situations is therefore essential for achieving any no net loss goal. In almost all situations – where multiple biodiversity features are included in a no net loss goal, over a broad landscape – options for offsetting are sufficiently complex that these upper limits can only realistically be defined by reference to systematic conservation plans that identify conservation priorities based on a clear societal biodiversity goal with specific targets.

Conditions required to improve upon the status quo	None related to upper limits to what impacts can or should be offset – properly implemented offsets will often improve upon the status quo.
Conditions required for no net loss	Upper limits defined – ideally by reference to systematic conservation plans that identify conservation priorities based on a clear societal biodiversity goal with specific targets.

5.2.6 Lower limits to what impacts can or should be offset

Offsets are least critical as a strategy for low significance impacts, e.g. small impacts on biodiversity of low irreplaceability and low vulnerability. High significance impacts (e.g. large impacts on rare biodiversity) will generally be most difficult and risky to offset. Offsetting is best placed to address medium significance impacts, such as sizeable impacts on common or widespread biodiversity or small impacts on rarer biodiversity (Pilgrim et al., 2013b; Regnery et al., 2013). A 'lower limit' for offsets can thus be envisaged, below which it is not as relevant, practical or efficient to compensate for impacts in the same way as it would be for impacts of greater conservation concern, i.e. where high offset transaction costs would be incurred for limited biodiversity benefits (e.g. as suggested for Great Crested Newts in the UK: Dickie & Tucker, 2010). Exactly where such lower limits should be drawn depends on national or sub-national biodiversity goals. In order to halt biodiversity decline or restore biodiversity towards policy targets, some policy-makers might find it necessary to require offsets for all residual impacts on biodiversity that is at all vulnerable or irreplaceable. This may particularly be the case in jurisdictions where biodiversity is declining rapidly in quality or extent (or where biodiversity has already declined considerably). The broader the policy goals, the more likely that this will be the case: for example, progressively lower thresholds would likely be set for offsets if relevant biodiversity policies attempt to incorporate climate change adaptation opportunities, conserve genetic diversity, or ensure ecosystem service values of biodiversity are captured. Offsetting all small-scale impacts on biodiversity of least conservation concern would, however, only be practicable if a simpler offsetting system could be established – the only currently available system is of developer contributions or 'in-lieu' fees (Treweek et al., 2008) – or if minor impacts and/or impacts on biodiversity of least conservation concern were altogether excluded from requirements from compensation. This latter option is often the case in practice, i.e. a lower limit to required compensation is often circumscribed in existing policy or legislation. For example, the New Zealand Resource Management Act requires compensation only for significant impacts (although definition of 'significant' has been the subject of much discussion: e.g. Norton & Roper-Lindsay, 2004, 2008; Walker et al., 2008). Where lower limits to impacts requiring compensation exist in legislation, however, such impacts are usually not even recorded (e.g. in environmental impact assessments).

Summary of approaches to lower limits to what impacts can or should be offset:

- no lower limits defined;
- lower limits defined, with compensation (but not full offsets) required below such limits; and
- lower limits defined, with no compensation required below such limits.

Conditions required to improve upon the status quo or achieve no net loss:

Given a status quo of limited or inadequate compensation for development project impacts, any offsetting will be an improvement – however much is excluded by any lower limit to required offsets. An offset system would, however, be most efficient if lower limits to what impacts can or should be offset are clearly defined (for impacts on biodiversity that are of least conservation concern), below which a simpler impact compensation system (such as in lieu fees) is applied (Dickie & Tucker, 2010).

Whatever the definition of 'no net loss' in a particular situation, it can never include all biodiversity (much of which is still not known to science), so will always exclude some impacts. Lower limits to offsetability are not essential for achieving no net loss (although they would improve efficiency of any offset system, as described above). If, however, lower limits are described, no net loss clearly cannot be achieved unless all impacts/biodiversity within the relevant definition of no net loss fall above these lower limits (i.e. are included in an offset system).

Conditions required to improve upon the status quo	None related to lower limits to what impacts can or should be offset – properly implemented offsets will often improve upon the status quo.
Conditions required for no net loss	None. However, if they are defined, lower limits to what impacts can or should be offset must not exclude impacts/biodiversity within the relevant definition of 'no net loss'.
Additional notes	Lower limits to what impacts can or should be offset, below which a simpler compensation system applies, would make for the most efficient offset system.

5.2.7 Relative offsetability between lower and upper limits to what impacts can or should be offset

Guidance is necessary to ensure developers understand how to best design offsets and ensure regulators can transparently and consistently judge among development and offset proposals. The key feature of such guidance is a transparent system for assessment of relatively offsetability, i.e. considering the theoretical possibility (e.g., availability of suitable offset sites for a given biodiversity feature), practical achievability (e.g., availability of relevant proven offset delivery techniques) and appropriateness (e.g. due to risks to biodiversity) of offsets (Pilgrim et al., 2013a; Section 5.2.5). Pilgrim et al. (2013a) propose a framework which establishes the burden of proof necessary to confirm the offsetability of impacts, given varying levels of: conservation concern for affected biodiversity; residual impact magnitude; opportunity for suitable offsets; and feasibility of offset implementation in practice. This framework allows categorization of the relative burden on developers to demonstrate that there is limited danger to biodiversity in shifting from a lower-risk status quo to a new position (i.e. with development and offsets). Such a framework would promote development of offset proposals which are more demonstrably likely to be successful and/or would promote change to more offsetable development proposals. A number of the factors which would need to be considered in such a framework are discussed elsewhere in this report (e.g. temporal loss in Section 5.2.10, uncertainty and risk in Section 5.2.12).

Summary of approaches to relative offsetability:

- no guidance on relative offsetability; and
- guidance on relative offsetability.

Conditions required to improve upon the status quo or achieve no net loss:

Given a status quo of limited or inadequate compensation for development project impacts, any offsetting will be an improvement – however difficult it is to judge among benefits of varying offset proposals. Any guidance on relative offsetability would, however, improve offset design and decision-making. Theoretically, 'no net loss' could be achieved – particularly where definitions of no net loss are simple – without guidance on relative offsetability. The likelihood of achieving no net loss would, however, be significantly higher if such guidance exists, because it would permit more informed decision-making by developers and regulators.

Conditions required to improve upon the status quo	None related to guidance on relative offsetability – properly implemented offsets will often improve upon the status quo.
Conditions required for no net loss	Guidance on relative offsetability is not a pre-requisite for achievement of no net loss, but would be a significant factor in likelihood of success of no net loss programmes.

5.2.8 Additionality

'Additionality' is a generally agreed principle of offsetting, referring to the need for offsets to provide a new contribution to conservation, beyond a counterfactual scenario (e.g. business as usual) (e.g. ICMM, 2005; McKenney & Kiesecker, 2010; BBOP, 2012). Counterfactuals are, however, difficult to estimate with certainty – particularly with regard to averted loss: the REDD carbon experience has demonstrated how difficult it is to prove that an area would have been cleared had economic incentives not been available. Maron et al. (2013) found limited attention to establishing and measuring counterfactuals in offset literature and policy, and highlighted the critical need to calculate expected gains from offsets against realistic and explicit counterfactuals – a recommendation reinforced by modelling of various counterfactuals by Bull et al. (2014).

Counterfactuals are easier to establish with confidence in some cases than others. Maron et al. (2012) consider that "there are limited circumstances under which averted loss can be considered true additionality (particularly in nations with well-developed biodiversity protection controls)". The prevalence of preservation (i.e. averted loss) as a strategy within species-banking in the USA is seen by Pawliczek & Sullivan (2011) as indicating limited additionality. Fox & Nino-Murcia (2005) estimated that 49% of species banks surveyed exhibited additionality, i.e. "would most likely have been destroyed or seriously degraded by competing land uses if banking had not been an option". Additionality is usually easier to infer or prove for restoration – rather than averted loss – offsets, since restoration is usually the result of concerted efforts by, or coordinated by, a single stakeholder in an already degraded area, whereas ongoing biodiversity loss (to be averted) is often caused by a diverse set of actions by multiple stakeholders. However, there are certainly cases where offsets may take place at sites where voluntary restoration efforts (e.g. by conservation NGOs) would otherwise have occurred, or where governments decrease conservation budgets in response to new offset funding streams for conservation (Overton et al., 2012). These cases of non-additionality are even more difficult to predict than cases where development would not have occurred. Understanding of additionality will improve with increased and better-quantified knowledge of the conditions under which protected areas yield biodiversity benefits, as per the first objective of the IUCN WCPA/SSC Joint Task Force on Biodiversity and Protected Areas.

Even where preservation of a truly threatened site is genuinely over and above business as usual, additionality can be compromised by leakage of those threats to other areas (Ewers & Rodrigues, 2008), a factor that could be addressed in forest-based carbon markets with comprehensive global incentives (Strassburg et al., 2009) but in practice is being dealt with in voluntary markets by percentage carbon leakage credit adjustments based on broad categories of risk (VCS, 2008). These credit adjustments reduce the value of carbon credits on a percentage basis, to a greater degree for higher-risk projects. For example, a forest protection project calculated to produce 200 Verified Carbon Units (VCUs) may be required to reduce the value of the carbon credits they can sell by 70% (to 60 VCUs) if it is considered that timber harvest for newly-protected forests may leak to more carbon dense forests in the same country, or by 40% (to 120 VCUs) if such leakage is likely to similarly carbon dense forests in the country (VCS, 2008). A further, little-discussed, issue with requirements for additionality is that they often provide perverse disincentives for ongoing good stewardship of biodiverse land: landowners who have conserved biodiversity are not usually rewarded by an offsetting scheme as their ongoing efforts are not seen as additional until they degrade, or threaten to degrade, their land. This is particularly the case during the period preceding baseline definition (Dickie & Tucker, 2010).

The complexities of proving additionality in carbon markets led the Chicago Climate Exchange (in contrast to, e.g., the Clean Development Mechanism) to abandon rigorous criteria for additionality and instead pre-approve classes of project that have a high likelihood of additionality (Dickie & Tucker, 2010). The Chicago Climate Exchange found that rigorous additionality requirements significantly increased transaction costs, yet exposed few false positives (non-additionality) while potentially excluding a greater number of false negatives (projects which truly were additional yet could not prove it). To some degree, additionality can be introduced back into the market under a cap-and-trade system (Section 5.2.5) by allowing purchase and 'retirement' of offset credits by stakeholders other than developers (e.g. public agencies, NGOs: Dickie & Tucker, 2010).

Additionality is particularly difficult to demonstrate on land where there are existing legal obligations to conserve biodiversity – a notable problem in OECD countries where large proportions of valuable habitat are already under conservation or agri-environment schemes. It is particularly difficult to demonstrate additionality on state-managed land, especially for protected areas. In such areas, even where existing management is below that legally required, additionality – and thus offsets – may best be seen as encompassing solely management actions over and above those mandated by law (Dickie & Tucker, 2010; BBOP, 2012a). However, this issue remains controversial in non-OECD countries, with some stakeholders arguing that offsets should not be allowed in legally protected areas because they are non-additional (i.e. subject to existing government commitments) and other stakeholders arguing that such offsets are additional because non-OECD governments do not have the finances to support their existing commitments to protected area management. A compromise solution may be for long-term national protected area financing plans to be put in place, with external management funding to protected areas only considered additional (and thus equal to an offset) until the time at which the government could realistically fulfil its legal obligations. Such a solution may be reasonable, but could encourage uncommitted governments to repeatedly renege on obligations, knowing offset funding would fill budgetary gaps.

A further additionality consideration comes into play where 'credit stacking' is allowed – e.g. where a mitigation bank on a single site could unbundle multiple ecosystem functions (e.g. threatened species, wetland habitat, carbon sequestration, etc.) and sell them separately (Fox, 2008; Womble & Doyle, 2012). Although the policy rationale for such an approach is strong, by further incentivizing protection and restoration of habitat (Bekessy & Wintle, 2008; Bekessy et al., 2010), selling two credits for the same hectare has clear additionality problems and is administratively problematic because they may not be entirely independent. For instance, the key action required for one credit (e.g. protecting trees to sequester carbon) may also be the same required for another (e.g. preserving woodpeckers). Such non-additional use of credit stacking has been termed 'double-dipping'.

Summary of approaches to additionality:

- detailed additionality requirements;
- rule-of-thumb additionality requirements (such as the Chicago Climate Exchange);
- no additionality requirements;
- use, or not, of leakage credit adjustments;
- inclusion, or exclusion, of protected areas as additional in some way; and
- allowance, or not, of credit-stacking.

Conditions required to improve upon the status quo or achieve no net loss:

Even given a status quo of limited or inadequate compensation for development project impacts, offsetting will only be an improvement if it is additional to some extent. Rule-of-thumb additionality requirements are thus likely to be the minimum condition for offsetting to improve upon the status quo. The definition of 'no net loss' in a particular situation may to some extent define the level of detail to which additionality must be demonstrated, but clear additionality will be necessary to achieve no net loss – including a method such as credit adjustments to cope with leakage where necessary. It is likely that some offset activities in some protected areas would be viewed by most stakeholders as additional and therefore contributing to no net loss. The issue of credit-stacking, however, poses complex and as-yet unanswered questions.

Conditions required to improve upon the status quo	Rule-of-thumb additionality requirements are likely to be the minimum necessary to improve upon the status quo.
Conditions required for no net loss	Requirement for demonstration of clear additionality, plus methods such as credit adjustments to cope with leakage where necessary.
Additional notes	Considerably stronger stakeholder engagement would help move forward complex discussions on additionality of activities in protected areas and credit-stacking.

5.2.9 Permanence

Offset gains need to last at least as long as residual impacts. Given the permanence of many development footprints, at least a portion of offset gains usually need to be permanent (Gibbons & Lindenmayer, 2007; McKenney & Kiesecker, 2010). In most instances of permanent conservation, some permanent management – such as invasive species control – will be necessary (except for example, after complete removal of an invasive species from an island, or placement of a conservation easement on land that was previously allocated for development). Most offsets thus require both financial and legal mechanisms to assure gains remain permanent (where compensating for permanent impacts); they require: (i) assurance against unexpected disasters; (ii) funding for permanent management; and (iii) secure legal control over activities allowed in the area. These three mechanisms are clarified in the following three paragraphs.

Companies implementing voluntary offsets have generally been reluctant to provide up-front funding to assure long-term management, since business models and time preference (through standard financial discounting) promote provision of such funding at the time of closure. This leaves offsets vulnerable to business failure, corporate takeover, or natural or social disasters. However, potential solutions already exist and have been used extensively for other purposes – namely various forms of bonds, insurance and credit buffers. In regulated markets it is common practice to require provision of bonds or insurance to assure restoration upon closure (Section 5.2.12). Voluntary carbon markets also employ a form of insurance by withholding part of the emissions reduction credits from each forestry project, based on each project's potential for failure. These portions of credits are pooled together across all projects and this pool is used to replace any from individual projects that collapse (UNEP-FI, 2008). An alternative approach to such 'credit buffers' has been trialled on at least one occasion in biodiversity mitigation programmes: Wilkinson (2009) documented a case in South Carolina where a percentage of mitigation credit fees were earmarked for long-term management. In a similar fashion, Temple et al. (2012) explain how a mining project in Madagascar is aiming to achieve a very positive impact on biodiversity, in order to ensure a buffer such that a net positive impact can be achieved even in the case of partial offset failure. Overall, insurance, bonds and credit buffers present a highly promising approach to protecting against disaster and underachievement in biodiversity offsets.

In the many situations where offsets require active long-term management, insurance, bonds or credit buffers will be insufficient by themselves because they do not protect against slow loss of offset gains that may accumulate over time without effective management. In such cases, long-term financing mechanisms (e.g. trust funds) will also be required in order to generate year-on-year funding for management. Despite extensive use of long-term financing mechanisms for conservation, discussion of their potential for long-term management of REDD carbon offsets (Spergel & Wells, 2009), and recommendation of non-wasting endowments for long-term management under US conservation banking guidance, Wilkinson (2009) documented only two cases where long-term endowments had been established to support long-term management. Where long-term funding for management of offsets is not put in place, non-governmental or governmental bodies may take over management of offset sites once these sites no longer receive developer funding. While such adoption of management is beneficial in assuring permanence, it may mean that long-term offset gains are not truly additional because these bodies abandon other conservation plans in order to devote funds to managing offset sites for conservation.

Security of financing will only be effective if accompanied by assurance of management rights over land (e.g. through land title, easements, covenants, protected area designation, etc.: Gibbons & Lindenmayer, 2007; Gardner, 2008). For example, conservation banking guidance in the USA requires easements on land from which credits are traded (Madsen et al., 2010). In many legislatures (e.g. Canada, New Zealand), it is not possible to completely assure permanent protection of offset gains, owing to sub-surface rights (e.g. to exploit minerals) taking precedence over surface rights (e.g. land titles, easements) and no facility existing to retire these sub-surface rights, even in protected areas. In such situations, offset permanence cannot be assured because, for example, a mining company may later have rights to exploit sub-surface resources in a way that reduces offset gains[8].

Summary of approaches to permanence:

- no mechanisms to assure permanence;
- mechanisms to assure against disaster;
- mechanisms to assure long-term management; and
- mechanisms to secure land management rights.

Conditions required to improve upon the status quo or achieve no net loss:

Given a status quo of limited or inadequate compensation for development project impacts, any offsetting will be an improvement – for however long offsets last. However, no net loss cannot be achieved without mechanisms to assure permanence, including: (i) mechanisms to assure against disaster (e.g. insurance, bonds, or credit buffers); (ii) long-term financing mechanisms to assure long-term offset management (e.g. trust funds); and (iii) mechanisms for secure land control (such as legislation which enables retirement of all exploitation rights).

Conditions required to improve upon the status quo	None related to duration of offsets – properly implemented offsets will often improve upon the status quo.
Conditions required for no net loss	Mechanisms to assure permanence, such as retirement of all exploitation rights.

5.2.10 Consideration of temporal loss

Temporal loss of biodiversity (i.e. 'time lags') between development project impacts and full compensation through offset gains will raise risks of extinction to impacted biodiversity features. This is because those biodiversity features will suffer reduced viability (e.g. declines in extent, quality or density) for the time period between project impacts and offset gains. Such loss may be long-term. For example, Cunningham et al. (2007) found that planted vegetation was inferior habitat to remnant vegetation even after 20 years. Some key habitat features may take a very long time to develop (Wilkins et al., 2003; Maron et al., 2010), e.g. more than 500 years for ancient woodlands in Europe (Morris & Barham, 2007), and may thus potentially be considered non-offsetable (Treweek et al., 2010; Quétier & Lavorel, 2011). Lack of methods to deal with time lags will raise risks of extinction owing to habitat or population bottlenecks (Bedward et al., 2009; Bekessy et al., 2010; Maron et al., 2010), and thus potentially compromise achievement of offsetting goals.

Ideally, impacts of temporal loss would be considered through population viability analyses (Bonnie & Wilcove, 2008; Bruggeman & Jones, 2008; Bruggeman et al., 2009), or their correlates for other biodiversity features of importance such as habitats. Impacts/offsets would then need to be adjusted as appropriate if reduced viability compromised persistence of biodiversity to unacceptable levels, i.e. below certain thresholds (e.g. Swift & Hannon, 2010). However, such analyses are resource-intensive and hence only practical in highly-developed regulatory environments and/or for biodiversity of particularly high conservation concern (Bonnie & Wilcove, 2008).

The potential use of multipliers to address temporal loss is discussed in Section 5.2.11, with the conclusion that multipliers are inappropriate in many such situations. Theoretically, multipliers compensate for temporary losses by increasing future gains, but in most cases risks of extinction would be increased in the interim. Multipliers could, however, be an appropriate and simple strategy for dealing with human time preference when impacts on biodiversity are of low significance;

8 One can imagine an offsetting system that would, in such cases, require replacement offsets by those exploiting resources in an offset area, but at some point further offsets would not be possible owing to natural limits to the biodiversity in question having been reached (Section 5.2.5).

unaffected portions of affected biodiversity will remain viable and secure until the time at which offset gains are produced (Quétier & Lavorel, 2011). Time discounting, as often used within habitat equivalency analysis (Dunford et al., 2004; Moilanen et al., 2009), is essentially a particular way of calculating multipliers to address human time preference (Moilanen et al. also suggest that time discounting has a role to play in addressing uncertainty of offset success, but see Section 5.2.11). It should be noted that methods such as habitat equivalency analysis have potential not just to discount the value of future gains, but also to incorporate costs of biodiversity loss to humans on a year-on-year basis (e.g. 'discounted service acre years': NOAA-DARP, 1995). While generally applied for ecosystem services, there is no logical reason why such approaches should not be applied for existence values of biodiversity which are also cumulative year-on-year. For example, there is innate inequity if ten tigers are lost tomorrow and ten provided again in 50 years' time – not only is there time preference for the ten that exist today (suggesting that time discounting should be applied to existence values), but also the loss of those tigers is likely to be felt every year until they are reinstated (suggesting that cumulative year-on-year losses should be calculated). Indeed, such an application is suggested by the Environmental Liability Directive of the European Parliament (Directive 2004/35/CE; Lipton et al., 2008). Once again, however, it should be noted that time discounting and consideration of year-on-year losses alone will not solve raised extinction risks of temporal loss.

In most cases, rather than multipliers or population viability analysis, requiring production of certain (at least short-term) gains before impacts is the best way to avoid or reduce temporal loss, e.g. via species- or habitat-banking on a 'savings bank' basis (Walker et al., 2009; Bekessy et al., 2010; McKenney & Kiesecker, 2010; Quétier & Lavorel, 2011). 'Savings bank' approaches do, however, impose practical limitations on equivalence: when setting up offsets for banks, it is not always easy to predict what kinds of biodiversity may be impacted in the future. One approach would be to prioritize incorporation of the most significant sites for persistence of biodiversity, e.g. Key Biodiversity Areas (TBC 2012). A lack of banks containing sufficiently comparable biodiversity to allow a developer to buy credits may prevent development within 'savings bank' offset systems.

Summary of approaches to temporal loss:

- no attempt to deal with temporal loss;
- attempts to assess temporal loss using viability analyses;
- attempts to deal with temporal loss using multipliers; and
- avoidance or reduction of temporal loss using habitat/species savings banks.

Conditions required to improve upon the status quo or achieve no net loss:

Given a status quo of limited or inadequate compensation for development project impacts, any offsetting will be an improvement – whether, or however, time preference is accounted for. In some cases, 'no net loss' could be achieved without any attempt to deal with temporal loss. In most cases, however, extinction risks will be raised by temporal loss and thus the likelihood of achieving no net loss would be significantly higher if there is a requirement to produce offset gains prior to impacts (e.g. through habitat- or species-banking). In any cases where human time preference is intrinsic to definitions of no net loss, such savings bank approaches (or, for impacts of low significance to biodiversity, multipliers) will also be essential. In ideal scenarios, population viability analyses would be used to assess temporal loss of biodiversity, and all offset gains produced prior to impacts.

Conditions required to improve upon the status quo	None related to methods to address temporal loss – properly implemented offsets will often improve upon the status quo.
Conditions required for no net loss	Where no net loss definitions include human time preference, avoidance or reduction of temporal loss through up-front habitat-/species-banking (or, for impacts of relatively low significance on secure biodiversity, multipliers) will be necessary. Otherwise, savings bank approaches are not a pre-requisite for, but would be a significant factor in increasing the likelihood of achieving, no net loss.
Additional notes	Avoidance or reduction of temporal loss through up-front habitat-/species-banking (if population viability analyses show unacceptably reduced persistence).

5.2.11 Multipliers

Regulators may sometimes require *a priori* offset multipliers[9], i.e. greater than one unit of biodiversity to be offset for each one impacted. Such a requirement may be based on an attempt to deal with various factors (e.g. Overton *et al.* 2012), including:

i. **achievement of biodiversity conservation goals** (e.g. Brownlie & Botha, 2009; DEA & DP, 2011). Multipliers are most useful when trying to ensure no net loss, or achieve net gain, towards an overall biodiversity-focused policy goal (e.g. within a systematic conservation plan; Brownlie & Botha, 2009). For example, the proposed national policy statement on indigenous biodiversity in New Zealand[10] requires offset design to demonstrate that "it contributes to and complements biodiversity conservation priorities/goals at the landscape and national level." Multipliers for this purpose are sometimes referred to as 'end-game' or 'conservation outcome' multipliers (BBOP Multipliers Consultation Working Group, 2008; BBOP, 2012b). In such situations, regulators might require high multipliers for offsets that impact depleted biodiversity (e.g. a species' population that is already reduced to 20% of its original size, or a habitat that has already been reduced to 40% of its original extent) in an attempt to achieve net gain in that biodiversity (i.e. increasing their current extents/populations towards historical levels). Such an 'end-game' approach facilitates contribution to strategic goals (i.e. the ultimate desired conservation outcomes). Such an approach also steers geographically-flexible development away from areas with biodiversity of conservation concern (i.e. ones important to strategic goals) because of higher multipliers for (and thus costs of) offsetting in these areas.

ii. **lack of precision in available data/predictive power** (Burgin 2008; Brownlie & Botha, 2009; Moilanen et al., 2009; Treweek et al., 2010; Quétier & Lavorel, 2011). For example, the scale of residual impacts or of offset gains may be bounded by error margins owing to poor data quality or limited ability to precisely predict biodiversity responses to impacts/conservation measures. This lack of precision can reasonably be dealt with by multipliers that ensure precautionary estimates of losses and gains throughout, although the scale of such multipliers (given significant uncertainty over restoration of biodiversity) may be impractically large (Moilanen et al., 2009; Bekessy et al., 2010). Production of offset gains before development impacts (e.g. through conservation banking 'savings banks': Norton, 2009; Bekessy et al., 2010; but also see limitations – Section 5.2.10), or insurance or bonds would be other potential ways of addressing this issue (Gerard, 2000; Shogren et al., 1993; Miller, 2005; Baber, 2012; Maron et al., 2012). Insurance/bonds would have lower overall costs to developers (as they might not always be needed) but would have higher overall costs to biodiversity (because, when used, they would result in delayed fulfilment of required offset gains).

iii. **uncertainty of offset success**, i.e. whether an offset might fail altogether (e.g. due to unproven offset delivery techniques or commercial failure) rather than incompletely succeed (this latter is discussed in (ii), above, and, e.g., in Burgin, 2008; Moilanen et al., 2009; Overton et al., 2012). This is an inappropriate situation in which to use multipliers (Walker et al., 2009), because uncertainty of result is not always reduced by quantity (e.g. the chance of flipping a coin twice and getting a 'heads' the second time is not influenced by the result of the first flip). In fact, success of offsets is likely to be highly correlated across sites (Moilanen et al., 2009). Production of offset gains before development impacts (e.g. through conservation banking 'savings banks': Section 5.2.10) is thus the best method for dealing with uncertainty of offset success (or, where not possible, bet-hedging, insurance or bonds; Section 5.2.12).

iv. **temporal loss** (Brownlie & Botha, 2009; Moilanen et al., 2009). This has been proposed (e.g. Evans et al., 2013) but is often not an appropriate situation in which to use multipliers (Walker et al., 2009; Quétier & Lavorel, 2011) because, e.g., extinction risks of habitat bottlenecks are not reduced by requirements for greater amounts of habitat at a future time (the same applies to population bottlenecks). Multipliers (such as those calculated through a method of time discounting; Moilanen et al., 2009; Wissel & Wätzold, 2010; Evans et al., 2013) could, however, be appropriate for dealing with human time preference (e.g. Overton et al., 2012) when impacts on biodiversity

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9 Although 'offset multiplier' and 'offset ratio' are often used interchangeably, we confine use of the latter term to its original use (e.g. in US wetland mitigation banking: Stein et al., 2000; Bonnie & Wilcove, 2008) of being simply an observation of offset area divided by impact area.
10 http://www.mfe.govt.nz/publications/biodiversity/indigenous-biodiversity/index.html

are of low significance, i.e. unaffected portions of affected biodiversity will remain viable and secure until the time at which offset gains are produced. Production of certain (at least short-term) offset gains up-front is the best way to deal with temporal loss, e.g. via conservation banking 'savings banks' (Section 5.2.10).

v. **inadequate exchange currencies** (McKenney & Kiesecker, 2010). Multipliers are sometimes used by regulators to compensate for 'quality' issues not adequately captured in currencies. This is a coarse approach, and less beneficial than improving metrics or currencies.

When multipliers are calculated appropriately, i.e. to assess theoretical magnitudes necessary to address uncertainty or other factors, very high ratios (e.g. >100:1) may be required to guarantee no net loss (Moilanen et al., 2009; Pickett et al., 2013). Such ratios may deter development, but are an accurate reflection of the likelihood of potential losses versus gains.

Summary of approaches to use of multipliers:

- no use of offset multipliers;
- use of offset multipliers in inappropriate ways (to address uncertainty of offset success, to compensate for inadequate currencies, or to address raised extinction risks of temporal loss); and
- use of offset multipliers in appropriate ways (to take a precautionary approach to lack of precision, or to achieve conservation goals, or to address human time preference).

Conditions required to improve upon the status quo or achieve no net loss:

Given a status quo of limited or inadequate compensation for development project impacts, any offsetting will be an improvement – whether or not it still results in some losses to biodiversity through lack of multipliers or their inappropriate use. Achievement of 'no net loss' is likely to require use of multipliers to take a precautionary approach to lack of precision. If the definition of no net loss is to include human time preference (Section 5.2.10), it will, in some cases, be necessary to also require multipliers to address time preference for impacts of relatively low significance on secure biodiversity. This will not, however, address temporal loss – i.e. reduce extinction risks in the short-term for biodiversity of conservation concern that may have its viability reduced by time lags between development impacts and offset gains. In such cases, which are likely to be the norm (since offsetting was conceived in order to assist conservation efforts in the face of development), methods for addressing temporal loss will also be necessary (Section 5.2.10).

In an ideal offsetting system, use of multipliers to take a precautionary approach to lack of precision and to achieve conservation goals (and in some cases to address time preference) would be the approach that provided the optimal gains from offsetting.

Conditions required to improve upon the status quo	None related to presence or absence of multipliers, or the appropriateness of their use – properly implemented offsets will often improve upon the status quo.
Conditions required for no net loss	Use of multipliers to account for lack of precision. If 'no net loss' definitions include human time preference, multipliers or other approaches will be necessary for impacts of relatively low significance on secure biodiversity (but will not be sufficient to achieve no net loss for more significant impacts).
Additional notes	In an ideal offsetting system, use of multipliers to account for lack of precision and to achieve conservation goals (and in some cases to address time preference) would be the most productive approach.

5.2.12 Uncertainty and risk

All decision-making regarding offsets, from understanding of impacts to measurement of gains, will be constrained by the quality and quantity of relevant data available on biodiversity, impacts, restoration techniques, etc. (Burgin, 2008; BBOP, 2012b). Offset quality and relevance will thus be directly correlated to, and constrained by, availability and reliability of relevant data (Walker et al., 2009). The complexity of biodiversity inherently means that biodiversity-related data are always incomplete, and Walker et al. (2009) suggest that obtaining sufficient biodiversity data to inform exchanges usually exceed the resources that governments, developers or habitat bankers have been willing to fund. While approaches to modelling complex future scenarios are developing (e.g. Moilanen et al., 2005; Pouzols & Moilanen, 2013), a level of uncertainty will always exist in offsetting, resulting in a level of risk to biodiversity. A precautionary approach to offsetting is particularly key where uncertainty is high. Incorporation of risk measures into currencies would be ideal, but is impractical (Salzman & Ruhl, 2000).

Regan et al. (2002) explore the issue of uncertainty in conservation in depth, breaking it down into epistemic and linguistic uncertainty – to which Kujala et al. (2013) add human decision uncertainty. Varying sources of uncertainty will influence varying areas of offsetting. Inherently, uncertainty and risk cannot be managed well without comprehensive identification of sources of uncertainty and risk for a particular project, followed by estimation of the scale of uncertainties and estimation of the probability and consequences of risks. While some 'unknown unknowns' will always exist, estimation of the 'known unknowns' will facilitate an appropriate type and scale of management of uncertainty and risk. Bull et al. (2013) call for "…development of a comprehensive framework for treating uncertainty in offsets."

For the purposes of this report, we distil uncertainty in offsetting into three main types, focused on ultimate outcomes: (i) uncertainty over precision (e.g. of the exact quantity of residual impacts or offset gains); (ii) uncertainty over offset success (i.e. whether offsets will actually succeed in providing any gains at all); and (iii) uncertainty over whether offset gains can be sustained (i.e. whether gains that are provided can be sustained over time). The former two types of uncertainty pose two types of risk, respectively: the risk of insufficient offset gains to compensate for losses, and the risk of no offset gains to compensate for losses. In both cases, the optimal method to reduce or avoid uncertainty is to produce offset gains before development impacts, e.g. through conservation banking 'savings banks' (Section 5.2.10). These two types of uncertainty are discussed in Section 5.2.11 (ii) and (iii), with multipliers highlighted as appropriate for the former (although they may be impractically large: Moilanen et al., 2009; Bekessy et al., 2010; Pickett et al., 2013). Rather than multipliers, bet-hedging, insurance or bonds are more appropriate for addressing uncertainty of offset success (Gerard, 2000; Shogren et al., 1993; Miller, 2005; Burgin, 2008; Moilanen et al., 2009; Maron et al., 2012). Bet-hedging is simply the approach of selecting a portfolio of different offset areas that receive varying appropriate conservation interventions, reducing the chances of complete failure – particularly over all biodiversity components (Moilanen et al., 2009). As well as uncertainty of success, insurance and bonds are most appropriate for addressing uncertainty over sustaining offset gains in perpetuity (Section 5.2.9). Although insurance/bonds are a relatively new idea in offsetting, they are commonly used to ensure that mining companies can guarantee rehabilitation upon closure, even if the company collapses beforehand (e.g. Gerard, 2000; Shogren et al., 1993; Miller, 2005; Baber, 2012). An example of a bond being required for insurance of success of offsets is for impacts on seagrass in New South Wales[11].While uncertainty is clearly related to data availability and quality (Walker et al., 2009), Norton (2009) notes that uncertainty also generally increases with the intensity of restoration planned – i.e., restoration of highly-modified areas is generally more uncertain than restoration of slightly-modified areas. Uncertainty is also generally greater with regard to indirect (or 'secondary') impacts, rather than direct impacts: both direct and indirect impacts need to be considered for mitigation, including offsets (BBOP, 2012a; Gardner et al., 2013), as to some extent do cumulative impacts (BBOP, 2012a), but the nature, scale and duration of indirect impacts are more difficult to predict with a high degree of certainty (EBI, 2003).

Summary of approaches to address uncertainty and risk:

- no attempt to address uncertainty/risk;
- use of offset multipliers to address uncertainty/risk;
- use of bet-hedging to address uncertainty/risk;
- use of insurance/bonds to address uncertainty/risk; and
- production of offset gains before development impacts (e.g. via conservation banking 'savings banks').

Conditions required to improve upon the status quo or achieve no net loss:

Given a status quo of limited or inadequate compensation for development project impacts, any offsetting will be an improvement – whether or not there is uncertainty over the degree to which offsets will be successful, or even over whether they will succeed at all, and whether or not any attempt is made to address such uncertainty and risk. No net loss could certainly be achieved in some cases without consideration of uncertainty and risk, but systematic achievement of no

11 http://www.environment.nsw.gov.au/resources/greenoffsets/greenoffsets.pdf; http://www.dpi.nsw.gov.au/__data/assets/pdf_file/0019/202744/Fish-habitat-protection-plan-2---Seagrass.pdf

net loss will require uncertainty and risk to be addressed. Uncertainty over precision could be addressed by multipliers, insurance/bonds or production of offset gains before impacts. Uncertainty over offset success could be addressed by bet-hedging, insurance/bonds or production of offset gains before impacts. Uncertainty over whether offset gains can be sustained could be addressed by insurance/bonds. The approach that provides the optimal gains from offsetting would be production of offset gains before development impacts, e.g. through conservation savings banks, accompanied by insurance/bonds to assure long-term maintenance of gains.

Conditions required to improve upon the status quo	None related to uncertainty or whether/how it is addressed – properly implemented offsets will often improve upon the status quo.
Conditions required for no net loss	Uncertainty over precision: multipliers, insurance/bonds or production of offset gains before impacts. Uncertainty over offset success: bet-hedging, insurance/bonds or production of offset gains before impacts. Uncertainty over whether offset gains can be sustained: insurance/bonds.
Additional notes	Optimal offsetting would result from production of offset gains before development impacts, accompanied by insurance/bonds to assure long-term maintenance of gains.

5.3 Implementation conditions

Most of the conditions discussed in this section are generic conditions required for best achievement of most conservation-related activity (e.g., protected areas, community-based conservation), not just for offsets. As such, these are conditions that the conservation community has grappled with at length and there are few simple solutions. This report is focused on technical design aspects of offsets, and only attempts to give an overview of four implementation issues: (i) regulatory clarity; (ii) technical and financial capacity; (iii) free and transparent markets and oversight; and (iv) stakeholder engagement. In a companion policy input paper, ten Kate & Crowe (2014) discuss these issues in greater depth.

5.3.1 Regulatory clarity

Weak and ambiguous regulation is the norm for environmental legislation – an entrenched problem recognized by political scientists (e.g., Edelman, 1960; Section 5.3.3).There have been many calls for greater regulatory clarity – including on many of the technical issues in Section 5.2, but also on more practical issues (e.g. BenDor & Riggsbee, 2011; Clare et al., 2011) – but it has also frequently been argued that ensuring adherence to regulations (Sections 5.3.2, 5.3.3) is a more urgent need owing to poor or erratic implementation (Hough & Robertson, 2009; Quétier et al., 2013; Brown et al., 2014; Vaissière et al., 2014). A previous report (ICMM & IUCN, 2012) stresses the need for more practical experience, rather than theory, concluding that "Lessons learned from a community of practice will do more to further offset success than 10 years of theoretical debate." The issue of clarity in regulations and policy is addressed in more detail in a companion input paper (ten Kate & Crowe, 2014).

Conditions required to improve upon the status quo	In jurisdictions with regulations, greater clarity may be important, but more so may be enforcement of regulations and lessons learned from implementation.

5.3.2 Technical and financial capacity

For implementation:

Both companies and government departments are likely to want to implement offsets. Technical biodiversity capacity within both of these types of developer is often limited, particularly within companies and especially in relatively novel fields such as biodiversity offsetting. Developers are thus likely to have to outsource expertise. Unfortunately, the large international consultancy companies are also lacking expertise in these niche issues (ICMM & IUCN, 2012), and non-governmental organizations generally find it difficult to adapt swiftly to business needs at scale. Expertise in, and capacity for, design and implementation of biodiversity offsets is thus expected to be a significant need in the future. In regulated markets, such as the USA and Australia, entrepreneurs and small niche businesses are successfully filling this need. Biodiversity conservation – the fundamental underpinning of offset gains – is, however, extremely difficult in many countries. In particular, this is because conservation usually involves changing the behaviour of humans that are currently causing threats to biodiversity. Long-term stakeholder engagement will be key to successful offsetting, particularly in countries with less developed legislation or enforcement.

Financial capacity for offsetting should not be limited within government, and need not be limited within extractive industry: impacts are generally quite limited in relation to profits (compared to, for example, logging or agricultural industries, which operate over large areas with extensive impacts and lower profit margins). There is potential for regulatory requirements for offsets to raise costs to levels which prevent developments going ahead, but in practice this either does not appear to have arisen or developers have negotiated reductions in (or exclusions from) requirements from regulators. Perhaps owing to the novelty of biodiversity offsets, most developers have so far not resourced them at levels commensurate with traditional activities such as mitigation, rehabilitation or health and safety. A particular issue is reconciliation of long-term offset management funding needs (Section 5.2.9) with short corporate or government budgeting cycles. Nonetheless, there is a long history of developers providing socially-oriented trust funds and bonds/insurance for rehabilitation, so such an issue is far from insurmountable.

For review/monitoring/enforcement:

Plenty of guidance exists on implementation of regulatory offsets, and guidance is starting to emerge for offsets more broadly (e.g. BBOP, 2012a). Such guidance largely addresses technical issues related to offsetting (Section 5.2). However, in regulatory systems, where most offsetting experience exists to date, offsets have most often failed at the implementation stage – apparently due largely to insufficient attention to monitoring, oversight and enforcement (though other reasons appear to include a lack of feasibility testing and stakeholder engagement during overly theoretical design stages). Here, then, remain some of the biggest challenges for biodiversity offsetting.

Assessments of offset programmes, particularly mitigation banking, have found high rates of non-compliance, often greater than 50%, and inadequate investment of time and resources for monitoring by relevant institutions – reducing incentives for compliance and thus reducing the likelihood that offsets achieve stated goals (Harper & Quigley 2005; Quigley & Harper, 2006; Burgin, 2008; Matthews & Endress, 2008; Brownlie & Botha, 2009; Norton, 2009; Walker et al., 2009; Burgin, 2010; Wissel & Wätzold, 2010; Brown et al., 2013; Quétier et al., 2013). Greater capacity is likely needed in many relevant institutions, including environment departments, regulators and the judiciary – not only to monitor and enforce offset regulations, but also to more thoroughly review and ensure modification or rejection of unsuitable/impractical offset proposals in the first place (Walker et al., 2009; Brown et al., 2013). In France, Quétier et al. (2013) noted that the burden of designing and building adequate institutional arrangements has been shifted down to local and regional permitting authorities, and even developers themselves. Evidence from wetlands shows that restoration projects are far more likely to be compliant with performance standards than creation projects (Quigley & Harper, 2006; Kozich & Halvorsen, 2012). The relative success of habitat translocation remains unclear (Box, 2014). Offset success – ultimately more important than regulatory compliance – is, however, even less likely than compliance (Section 5.2.4) and varies significantly with achievability and appropriateness of goals, and the clarity with which these are defined (Matthews & Endress, 2008). In New Zealand, local authority monitoring costs can be built into consent conditions, but this mechanism is underutilized because monitoring is perceived as a low priority and potentially negative interface with developers (Baber 2012). Provisions for partial cover of government monitoring costs by developers also exist in some Australian offsetting systems (Treweek, 2009).

Some of the biggest challenges for application of offsets are thus likely to be ensuring that: (i) enforceability is built into offset proposals (Norton, 2009); (ii) consenting authorities have appropriate expertise to review and monitor offset projects (Norton, 2009; Baber, 2012) – preferably with independent oversight (Bekessy et al., 2010; Treweek et al., 2010) – into the long term (Levrel et al., 2012); and (iii) there are sufficient sanctions for non-compliance (Clare et al., 2011). The amount of review and restrictions in a free market are, however, inversely related to the amount of possible trade, so there are considerable trade-offs in developing a system that functions well yet produces the best results for biodiversity (Walker et al., 2009; Wissel & Wätzold, 2010; Womble & Doyle, 2012).

Conditions required to improve upon the status quo	Offsetting will likely improve upon a status quo of limited or inadequate compensation regardless of capacity available for review,

	implementation, monitoring and enforcement.
Conditions required for no net loss	Sufficient capacity to review, implement, and monitor offsets, and to enforce regulations.

5.3.3 Free and transparent markets and oversight

Markets are not solely relevant to conservation banking approaches, but to any situation in which trade occurs – i.e. to all but purely developer-implemented offsets (a very small proportion). Walker et al. (2009) point out that the public choice theory of politics predicts that private interests (e.g. developers) will defeat public interests (e.g. biodiversity conservation). A number of factors come into play in such predictions, including disincentives for biodiversity traders (buyers and sellers) to support a robust, meaningful exchange system, incentives for a non-precautionary approach (e.g. developers underestimating impacts – see also Fox & Nino-Murcia, 2005 and Clare et al., 2011), and conflicts of interest within regulatory departments (e.g. financial or political incentives for regulatory officials to give developers weak oversight and to approve simple but crude exchanges – Brownlie & Botha, 2009, BenDor & Riggsbee, 2011, Clare et al., 2011 and Greenwald et al., 2012 discuss similar conflicts). Walker et al. (2009) also note that information asymmetry is inherent in most offset systems – a problem whereby 'insiders' (regulators and traders) know more than 'outsiders' (biodiversity conservation interests and the public), and thus transparency in exchange is limited (e.g. judging whether an offset is equivalent). This latter point is also highlighted by Pawliczek & Sullivan (2011), who note that private (rather than transparent public) exchanges prevail in species-banking in the USA. Levrel et al. (2012) note that distortion in exchanges also occurs because of legal preference for precedent, rather than case-specific solutions.

Most of the issues raised above, although clearly problematic, are not ones that mean offsetting, however poorly implemented, would not improve a status quo of development without compensation. Walker et al. (2009) do, however, highlight two situations in which offsetting might produce worse results than this status quo. First, when offsets supersede existing protection legislation, as happened with introduction of the Habitat Conservation Plan (enabling offsetting) to the United States Endangered Species Act, eroding the previous absolute prohibition on impacts leading to loss of individuals of endangered species. Nonetheless, the previous absolute prohibition had produced perverse incentives for landowners to 'shoot, shovel and shut up', whereas offsets have produced economic incentives for the same landowners to conserve their biodiversity (Pawliczek & Sullivan, 2011). Effective biodiversity protection, despite good regulations, is in any case rare in the face of development pressure – even in high-capacity countries such as the UK (Treweek, 2009). Second, in countries with active stakeholder engagement in development decisions, offset policies may remove or reduce environmental concerns yet be largely symbolic and thus fail to achieve positive biodiversity outcomes (Edelman, 1960; Walker et al., 2009; Gardner et al., 2013; Quétier et al., 2013).

Conditions required to improve upon the status quo	Offsetting will likely improve a status quo of limited or inadequate compensation in any jurisdictions which currently do not have well-developed biodiversity protection or stakeholder engagement in development decisions. Where well-developed biodiversity protection exists, offsetting may not always be appropriate. Where stakeholder engagement is limited, a transparent oversight process will best address issues.

5.3.4 Stakeholder engagement

A plan that has greater stakeholder and rightsholder buy-in and clarity is more likely to be implemented: there is great value to involving development, government and conservation stakeholders in the preparation of a conservation plan as a spatial framework for offsetting (ICMM, 2005; Kiesecker et al., 2009; Underwood, 2011). That is not to suggest that stakeholder engagement will necessarily be easy (it will usually be complex and time-consuming) or straightforward (many stakeholders may oppose development), but that time spent on transparency, stakeholder engagement and consensus-building in the short-term may often save significantly greater time later on issues such as permitting delays, additional permitting conditions, protests and complaints and lawsuits. Stakeholder and rightsholder engagement will be particularly important in identifying biodiversity conservation goals (e.g. through systematic conservation planning: ICMM, 2005; Clare et al., 2011) and developing exchange rules that reflect societal preferences – Temple et al. (2010) document an example of consultation resulting in trading up of compensation among habitats in the UK, while Hajkowicz & Collins (2009) demonstrate how metrics can be weighted by stakeholder preferences. Clare *et al.* (2011) also point out the potential value of involving local people in monitoring compliance. Significant guidance on stakeholder engagement already exists in both the mining and conservation sectors (ICMM & IUCN, 2012).

'No net loss' goals are inherently expressions of societal desire (e.g. Sections 5.1.1 and 5.2.5), and so achievement of no net loss will fundamentally require stakeholder engagement during identification of scope, scale and location of offsetting, and especially in development of the rules of exchange mechanisms. The conservation community will need to proactively engage in offsetting discussions, as in other development decisions, with constructive criticism intended to improve biodiversity outcomes. Such criticism is likely to involve a range of approaches (Robinson, 2012), but positive collaboration within safe learning environments is likely to be critical. For example, ICMM & IUCN (2012) concluded that "Business remains hesitant to invest in offsets due to uncertainty of the outcome as a risk-management tool" – i.e. that a perceived lack of scientific/stakeholder consensus on offsets suggests to companies that any voluntary offset efforts will not receive broad stakeholder acclaim.

Conditions required to improve upon the status quo	None - any level of stakeholder engagement in offsetting will often improve a status quo of limited or inadequate engagement in development decisions.
Conditions required for no net loss	'No net loss' goals are usually an expression of societal desires, and so stakeholder engagement is needed during identification of scope, scale and location of offsetting, and in development of exchange rules.

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