Study on specific design elements of biodiversity offsets: Biodiversity metrics and mechanisms for securing long term conservation benefits

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Final Report

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Final Report

A report submitted by ICF Consulting Services
in association with IEEP and associated experts

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Abstract

Biodiversity offsets have been identified as an essential component of an EU No Net Loss initiative. Offsets will contribute to achieving the EU's strategy to halt the loss of biodiversity by 2020, providing they are carefully designed to achieve measurable and sufficient conservation benefits and to maintain these in the long term. This study researches the requirements and options of specific design elements of biodiversity offsets, with a view to implementing and operationalising the EU No Net Loss initiative by 2015. It reviews international best practice of designing offset metrics and establishing mechanisms for ensuring long term conservation benefits and explores the implementation issues that could be faced in the EU. It is clear that different offset metrics and different combinations of mechanisms will be appropriate in different EU countries and in different situations and locations. As such an EU offset policy should allow for a balance to be struck between systems that are suitably prescriptive to establish common minimum standards for maintenance of long-term benefits, and systems that are realistic and achievable and can be maintained over time.

Executive Summary

Introduction

ICF International, the Institute for European Environmental Policy (IEEP) and associated experts were commissioned by the European Commission, DG Environment, to undertake a study to provide guidance on specific design elements of biodiversity offsets, with a view to implementing and operationalising the EU No Net Loss initiative by 2015.

The EU's biodiversity strategy specifies a target to halt the loss of biodiversity by 2020. A No Net Loss initiative is an important component of the EU's strategy to achieve this. In accordance with the mitigation hierarchy, it will seek to strengthen efforts to first avoid, then minimise, then restore losses of biodiversity and ecosystem services, and to compensate for any residual losses. The principle of no net loss recognises that, because of the variety of pressures facing biodiversity in the EU, even with renewed efforts to strengthen the avoidance, minimisation and restoration of impacts on biodiversity, the absolute prevention of biodiversity loss is unlikely to be achievable, and some residual impacts will inevitably occur. Biodiversity offsets – which will deliver conservation gains to counterbalance these residual losses - can therefore be an essential component of an EU NNL initiative.

Biodiversity offsets are defined by the Business and Biodiversity Offsets Program as:

> *Measurable conservation outcomes resulting from actions designed to compensate for significant residual adverse biodiversity impacts arising from project development after appropriate prevention and mitigation measures have been taken. The goal of biodiversity offsets is to achieve no net loss and preferably a net gain of biodiversity on the ground with respect to species composition, habitat structure and ecosystem function and people's use and cultural values associated with biodiversity.*

Previous research has identified two main types of design criteria that biodiversity offsets must address if they are to achieve no net loss. These criteria relate to the definition of offset requirements (i.e. the extent and type of offset needed), and to arrangements for implementing offsets and habitat banking (i.e. the system that is put in place to ensure that offsets achieve their intended results). This study has provided an opportunity to improve our understanding of these two aspects of offsets design, and how they might work in the EU. It examined:

- The metrics used to determine offset requirements for biodiversity and ecosystem services; and
- The mechanisms used to secure long term conservation benefits

These two elements are important in the design of any offset or compensation scheme, and need to be considered when applying offsets under existing rules (such as any compensation for impacts under the Habitats and Environmental Liability Directives) as well as any new offset arrangements developed through the NNL initiative.

The research reviewed experience in relation to these two elements, assessed their application to the EU context, and presents possible options for implementation in the EU.

While the work focussed on offsets, it is to be seen within the wider context of the No Net Loss initiative, and as one of the steps in the mitigation hierarchy. The focus is on the practical issues that would need to be addressed if offsets were to be promoted across the EU, rather than the advantages and disadvantages of offsets as a policy option.

Metrics

Offsets aim to ensure no net loss (or a net gain) of biodiversity and this should apply to all components of biodiversity importance that are significantly impacted. To ensure this objective is achieved it is necessary to measure biodiversity losses from impacts (biodiversity debits) and the gains from offsets (biodiversity credits) in a practical and transparent way so that their equivalency can be compared.

However, measuring losses and gains in biodiversity and ecosystem services is not straightforward due to their complexity and context related variability. To overcome problems of measurement, metrics are used that define common currencies[1] and units of biodiversity so that the amount of biodiversity loss from impacts and the amount gained from offsets can be quantified and compared to establish if no net loss, or net gain are achieved.

The varied incorporation and treatment of biodiversity properties and ecosystem services gives rise to a large number of metrics. As part of this study a simple typology was established that relates to the biodiversity components that are the primary focus of the metric (i.e. habitats or species) and the factors that are considered in assessing their ecological value. On this basis the following main types of metric can be identified (although there are many variations and overlaps):

- Habitat (biotope) area;
- Habitat (biotope) area x standard value;
- Habitat (biotope) area x site condition;
- Habitat (biotope) area x standard value x site condition;
- Species-focussed approaches;
- Habitat replacement costs;
- Ecosystem service specific metrics; and
- Economic valuation.

In addition to these listed metrics, expert judgement may sometimes be used along with stakeholder discussions and negotiations. It is also noted that, in practice, metric outputs may sometimes be used as an information source to inform negotiations between parties, rather than as a strict measure of offset requirements (especially where offsetting requirements are not clearly backed up by legal obligations.

It is common practice to adjust metrics using multipliers to address a number of issues that are not normally addressed within the metric. These typically include sources of risk and uncertainty, social equity and distributional issues, desires to ensure a particular long-term outcome from averted risk offsets, and to deal with losses that are temporary.

With the exception of the simplest ratio metrics, which are probably not fit for the purpose of a No Net Loss determination, all have their strengths and weaknesses and are suitable for use in some situations. In other words there is no single best metric or best-practice approach, and they need to be chosen according to their purpose, with reference to good practice principles that metrics should endeavour to incorporate (such as ensuring they result in equity in type, space and time of biodiversity and ecosystem services). This is crucial because the success of offsets is highly dependent on the use of appropriate metrics.

The use of basic biodiversity measures has led to one of the main criticisms of the offsetting approach. In this regard, habitat area ratio metrics which do not take into account of the value of habitats should generally be avoided because they are highly reductionist and are unlikely to be able to capture biodiversity values reliably. However they may still be suitable for the assessment of very low value habitats such as artificial habitats where it is more important to have low transaction costs so that workable offset schemes can be developed.

Our analysis indicates that the main approach to increasing the ability to capture key biodiversity values is to increase the sophistication of the metrics. This is in line with the scientific literature (e.g. Pereira et al. 2013). The use of sophisticated metrics has some drawbacks, including their reduced transparency, especially if numerous subjective or arbitrary judgements are required (e.g. on habitat classes and values, appropriate baselines / benchmarks for habitat condition and weighting factors). These issues can undermine confidence in the system amongst stakeholders. The more robust

[1] In this context 'currencies' are not concerned with placing a price on biodiversity components or ecosystem services.

metrics also need sufficient data, which often requires detailed and lengthy fieldwork by experts (especially if species are involved), which can have significant cost implications.

Because values associated with ecosystem services vary significantly from one site to another, establishing 'like-for-like' compensation for biodiversity offsetting is fraught with difficulty. Complex trade-offs are necessary where enhancing one service results in depletion of another. Local communities may be unwilling to accept offsets for biodiversity occurring away from the impact site if this entails the loss of locally valued ecosystem services. Ecosystem service metrics need to be chosen to ensure they are appropriate to the service and its context. However data requirements are likely to be high as several services, which may be location-specific, may need to be assessed each with a different metric, and data needs for each may be significant. The use of a variety of metrics may cause confusion. Further limitations may be encountered where economic valuation is desired.

The review clearly shows that whatever metrics are used they need to be carefully combined with appropriate exchange rules. This is important, because metrics do not capture all important biodiversity values and therefore a precautionary approach needs to be taken that guards exchanges in habitat type that could lead to undetected biodiversity losses. Thus exchange rules are needed to prevent high value habitats being replaced with lower value habitats (although exchanges within bands of similar value habitats may be appropriate) and areas of habitat being replaced by the same type of habitat but in lower condition (unless there is high probability that its condition will match the original habitat in a reasonable time).

Mechanisms for securing long term conservation benefits

While sound metrics are required to ensure that appropriate levels of conservation activity are specified, achieving no net loss also depends on these activities being delivered in an effective, sustained and measurable way over the long term. Securing long term conservation benefits from offset schemes relies on at least three main factors being satisfied:

- Ensuring the effective delivery of conservation management activities through appropriate regulatory and management systems;
- Securing the long term use of land for conservation purposes; and
- Ensuring the financial sustainability of conservation management over time.

Markets for conservation offsets around the world exist at different levels of maturity, and are influenced by very different institutional and geographical environments. As such they employ a variety of different mechanisms in different combinations in order to achieve a robust system that is compatible with local conditions.

Based on a review of international best practices one can synthesise that to provide secure long term conservation benefits, an offset must:

- **Be based on a binding contractual agreement** – i.e. the developer or provider makes a legally binding commitment to deliver the offset; this is a condition of the permit for the development; the contract/ permit specifies certain conditions that need to be complied with (e.g. regarding management actions, monitoring, reporting, financial aspects); and the regulator has the ability to enforce these conditions. The nature of the contract may vary according to the planning/ permitting/ regulatory structure in place.
- **Involve a long term management plan** – adherence to which is likely to be a condition of the contract. This will specify required actions, performance standards and targets, monitoring and reporting arrangements.
- **Secure rights to manage the land for conservation purposes**. This is most likely to be achieved through purchase of that land, although long term leases or long term management agreements specifying conservation actions are a possibility (with the proviso that they do not offer the same levels of long term security).
- **Involve obligations to use the land for conservation purposes in the long term / safeguards against changes in use**. This may involve a covenant or easement which specifies long term use, involvement of a 3rd party such as an NGO committed to conservation use, or long term

regulatory oversight / public scrutiny, perhaps backed up by information tools such as registers which specify that the land is to be used for conservation purposes.

- **Demonstrate secure access to finance** to fund conservation action. This will normally be achieved by requiring establishment of an appropriate conservation fund, though there may be alternatives (such as a bank guarantee).
- **Provide safeguards against risk of failure.** Such safeguards may be achieved through: metrics (e.g. the requirement for offsets includes a risk multiplier that allows for a certain % failure); regulatory measures (i.e. the regulator secures all reasonable safeguards); contingency funds (additional funds are added to allow for unforeseen costs); and/or financial insurance (insurance is provided against risk of technical or financial failure, perhaps through a collective pool into which all offset providers pay).

Policy Implications

Metrics

The principal conclusion from the analysis is that on the basis of the advantages and disadvantages of the various options for metrics, the best approach, in the short-term, would be to develop common principles on the development and use of metrics, perhaps accompanied by suggested important features of different metrics that could suit different situations.

Our recommendations on the key principles that could be further considered are:

- A set of metrics should be used that reflect the differing levels of importance of the various biodiversity and ecosystem services that are affected by impacts and the risks that offsetting residual impacts on them may result in uncompensated biodiversity losses.
- Multipliers should be used where necessary to adjust metrics according to potential risks of offset underperformance (and other uncertainties) and the need to ensure equitable outcomes and compensate for time delays in the provision of biodiversity gains from offsets.
- It is essential that metrics are used in conjunction with clear exchange rules that take a precautionary approach to ensuring no net loss (or agreed net gain objectives).
- The development and use of offset metrics needs to be underpinned by an appropriate policy and legislative framework, and adequate institutional support.

Mechanisms for securing long term conservation benefits

In order to secure long term conservation benefits, it is clear that different combinations of mechanisms will be appropriate for different countries, locations and situations. In designing an EU offset policy a balance must be found between systems that are suitably prescriptive to establish common minimum standards for maintenance of long-term benefits, and systems that are realistic and achievable, as well as those that can be maintained over time.

Whilst regulatory systems need to enforce ecological rigour, they also need to allow sufficient flexibility to ensure that offsets are viable ‘on the ground’. To achieve the correct balance between ecological rigour and flexibility of application is difficult even at the level of individual Member States: establishing detailed rules that can be applied consistently but flexibly across the 28 Member States of the European Union would be extremely challenging. However, a policy framework, setting out key principles for ensuring long term sustainability of conservation benefits could offer a more pragmatic way forward.

There are a number of options for ensuring security of offset land use, although they are not necessarily mutually exclusive and can be combined in different ways. The principal mechanisms include: land acquisition and leasing; management agreements; conservation covenants/easements; offset registries; site designation; and state and NGO stewardship. Whatever combinations are deemed most appropriate, it will be necessary to invest in creating an appropriate legal and institutional infrastructure that ensures that the mechanisms employed are mutually reinforcing.

Financial mechanisms to support the long-term delivery of conservation benefits from offsets can be divided thematically into 'mechanisms to ensure sufficient capital' and 'mechanisms to safeguard against risks of failure'. Conservation trust funds provide an internationally accepted means of financing offsets in the long term, and, other than the costs of financing them, there do not appear to be significant barriers to their application in the EU. Safeguards against risk can be secured through financial insurance, and/ or through the application of risk multipliers that increase offset requirements (and therefore allow a margin for failure). There is a need to examine insurance against risk across the system as a whole, and to avoid "over-insurance" which could entail excessive costs. As offsets are a new and complex prospect in many parts of the EU, requiring insurance against every potential delivery risk may add substantially to their overall costs. Endorsement of insurance pools may be a more desirable alternative to promote the early development of offset markets.

To achieve no net loss, offsets are required to deliver conservation benefits in perpetuity. However, even in the US and Australia, where offsetting experience is most established, offsets are relatively new and best practice is still emerging. We do not therefore have the experience to know how durable offsets are over the long term, and how well each of the identified mechanisms performs against its stated aims over the long term. There are examples in early US schemes where insufficient safeguards have resulted in failure to achieve no net loss. This has informed improvements in practice and the development of more stringent safeguards. However, the long term effectiveness of mechanisms available to secure long term conservation benefits cannot yet be fully evaluated. Ongoing monitoring and evaluation of these mechanisms is needed to assess the long term benefits of offsets, and hence their ability to achieve no net loss of biodiversity and ecosystem services.

1 Introduction

ICF International, the Institute for European Environmental Policy (IEEP) and associated experts were commissioned by the European Commission, DG Environment, to undertake a study to provide guidance on specific design elements of biodiversity offsets, with a view to implementing and operationalising the EU No Net Loss initiative by 2015. The study focused on two specific design elements:

- Metrics for assessing offset requirements; and
- Mechanisms for long term conservation benefits within the EU.

The research built on previous work for the Commission, including recent studies on design elements for habitat banking and policy options for delivering No Net Loss, and has sought to provide the Commission with more detailed guidance on two specific elements of biodiversity offsetting which are crucial for the future operationalisation of an EU No Net Loss scheme.

The research reviewed experience in relation to these two elements, assessed their application to the EU context, and presents possible options for implementation in the EU.

While the work focussed on offsets, it is to be seen within the wider context of the No Net Loss initiative, and as one of the steps in the mitigation hierarchy. The focus is on the practical issues that would need to be addressed if offsets were to be promoted across the EU, rather than the advantages and disadvantages of offsets as a policy option.

1.1 Overview of approach

The study involved the following tasks:

- Task 0 comprised an initial inception phase, including a kick off meeting and initial scoping work;
- Task 1 involved a review and analysis of biodiversity metrics;
- Task 2 involved a review and analysis of mechanisms for securing long term conservation benefits; and
- Task 3 examined options for the implementation of metrics and long term conservation mechanisms in the EU.

Figure 1.1 describes the overall approach to the study and the organisation of the work.

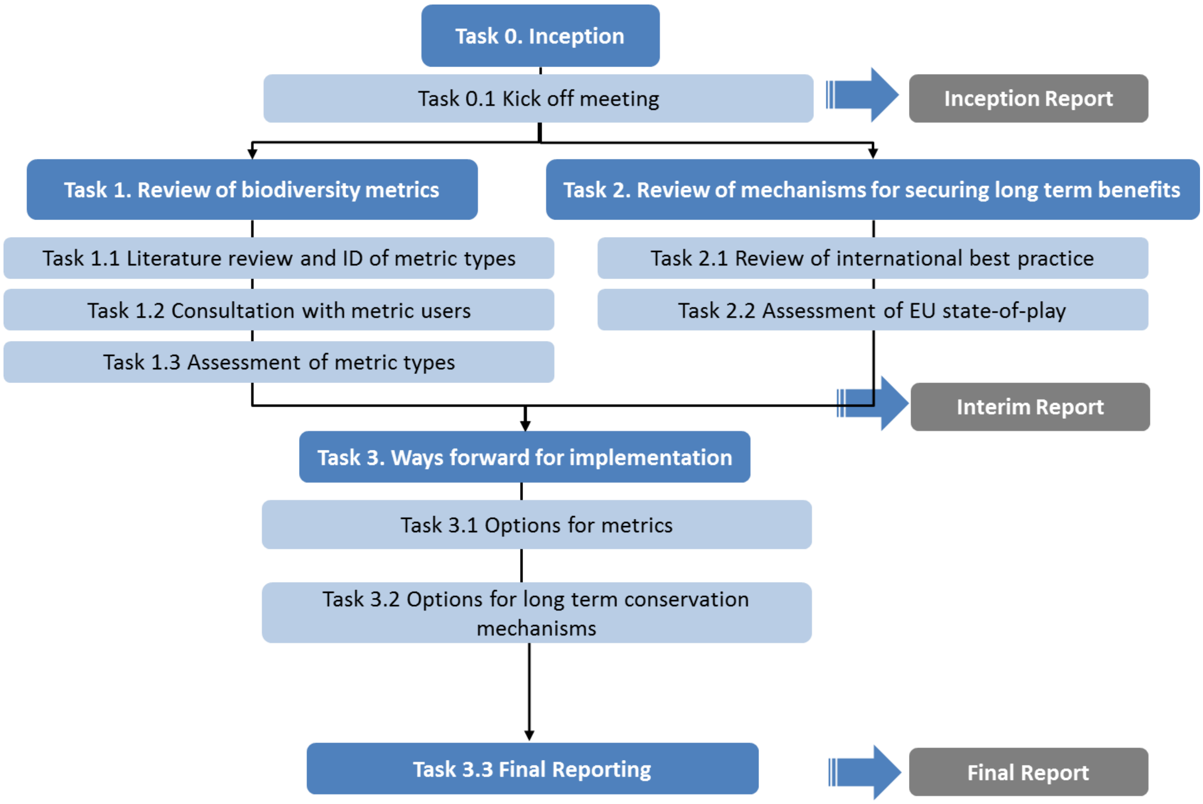
1.2 Structure of Report

This report presents the outputs of the research and is structured as follows:

- Section 2 presents a short overview of the policy context and the related objectives of this project.
- Section 3 presents a review and analysis of metrics for biodiversity offsetting and assesses their potential suitability for use under the EU NNL initiative.
- Section 4 addresses mechanisms for securing long term conservation benefits. It identifies the mechanisms currently employed as best practice in offsetting around the world and examines some of the issues that need to be considered in applying these in the EU.
- Section 5 draws out some of the principal options, in terms of metrics and mechanisms for securing long term benefits and assesses them against relevant criteria.
- Section 6 sets out the project conclusions and highlights areas for focussing future research.

- Annex 1: detailed reviews of metrics utilised in Australia, England, France, Germany South Africa and the USA

Figure 1.1 Overview of approach



2 Project Context and Objectives

The EU has a target to halt the loss of biodiversity by 2020

In 2011 the Commission launched "Our life insurance, our natural capital: an EU biodiversity strategy to 2020". The Strategy notes the continued and growing pressures on Europe's biodiversity, including land-use change, over-exploitation of biodiversity and its components, the spread of invasive alien species, pollution and climate change. It aims to reverse biodiversity loss and speed up the EU's transition towards a resource efficient and green economy.

The Strategy established the headline target of:

> *Halting the loss of biodiversity and the degradation of ecosystem services in the EU by 2020, and restoring them in so far as feasible, while stepping up the EU contribution to averting global biodiversity loss.*

A No Net Loss initiative is an important component of the EU's strategy to halt biodiversity loss

The 2020 Biodiversity Strategy established six more specific targets, and within them a set of 20 actions, which are together designed to achieve the headline target of halting biodiversity loss and the degradation of ecosystem services.

Target 2 relates to the maintenance and restoration of ecosystems and their services:

> *By 2020, ecosystems and their services are maintained and enhanced by establishing green infrastructure and restoring at least 15 % of degraded ecosystems.*

Within Target 2, Action 7 is to:

> *Ensure no net loss of biodiversity and ecosystem services.*

Action 7 is to be delivered through two more specific actions. The second of these (Action 7b) is that:

> *The Commission will carry out further work with a view to proposing by 2015 an initiative to ensure there is no net loss of ecosystems and their services (e.g. through compensation or offsetting schemes).*

The principle of no net loss recognises that, because of the variety of pressures facing biodiversity in the EU, while we can take measures to improve the avoidance of impacts on biodiversity, the absolute prevention of biodiversity loss is unlikely to be achievable, and that achieving no net loss – by requiring that residual losses are counterbalanced by equivalent gains – is a more realistic way of halting the overall loss of biodiversity and ecosystem services in the EU.

The concept of no net loss has been endorsed by the European Parliament which adopted a Resolution in April 2012 urging the Commission to develop an effective regulatory framework based on the 'No Net Loss' Initiative, taking into account the experience of Member States and the standards applied by the Business and Biodiversity Offsets Programme.

In order to advise on options for operationalising the concept, the Commission established a No Net Loss Working Group, involving a range of different stakeholder interests. The objective of the Working Group was to support the European Commission in its preparation of a NNL initiative by collecting views from Member State representatives, stakeholders and experts on the way forward.

The Commission also contracted IEEP, ICF and partners to undertake a study to support the development of the NNL initiative by defining and analysing alternative options that could achieve NNL at European level. The study has recently been completed (Tucker *et al*., 2014).

Biodiversity offsets are an essential component of an EU NNL initiative

The recent NNL options study highlighted the essential role of some form of biodiversity offsetting instrument in achieving no net loss of biodiversity and ecosystem services.

A wide range of instruments can potentially reduce rates of biodiversity loss, and move us towards no net loss. These instruments include nature protection laws, planning policies, pollution legislation, green taxes and a range of potential incentive measures. The NNL options study has found that applying, enforcing and extending existing policy instruments has an important role to play in achieving no net loss.

Indeed, the mitigation hierarchy requires that we avoid, minimise, and restore losses of biodiversity and ecosystem services, and that compensation for residual losses should be applied only as a last resort. However, recent evidence confirms that losses of biodiversity and ecosystem services are on-going and widespread. The need to promote growth and jobs is a continued priority at the EU level. Therefore, even with renewed conservation efforts, and improvements in the 'avoid, minimise and restore' steps of the mitigation hierarchy, some residual impacts will inevitably continue, particularly in cases where development and other pressures on biodiversity are considered to be inevitable and/or of over-riding public interest.

Achieving no net loss therefore will therefore require some mechanism to deliver gains in biodiversity to counterbalance these losses. Measures to address residual impacts will consequently be an essential part of a NNL policy.

In order to compensate effectively for residual impacts so as to achieve NNL, it will be necessary to be able to:

- Measure losses of biodiversity and ecosystem services;
- Design and implement measures that compensate for those losses with at least equivalent gains;
- Apply these measures to address all relevant impacts (including those caused by development, pollution, agricultural change and other pressures).

There is therefore a need for some form of arrangement for offsetting residual impacts on biodiversity and ecosystem services. Biodiversity offsets are defined by the Business and Biodiversity Offsets Program as:

> *Measurable conservation outcomes resulting from actions designed to compensate for significant residual adverse biodiversity impacts arising from project development after appropriate prevention and mitigation measures have been taken. The goal of biodiversity offsets is to achieve no net loss and preferably a net gain of biodiversity on the ground with respect to species composition, habitat structure and ecosystem function and people's use and cultural values associated with biodiversity.*

Previous research has identified two main types of design criteria that offsets must address if they are to achieve no net loss

Previous research for the European Commission by ICF GHK and BIO Intelligence Service examined the potential demand, supply, cost and design elements of biodiversity offsets, and the role of habitat banking schemes as an option for delivering them (ICF GHK, 2013). The report found that, if carefully designed and delivered, offsets have an important role to play in delivering no net loss of biodiversity and ecosystem services at EU level, but that this would depend on the development of an appropriate policy framework to require their uptake.

The research highlighted that two main types of design elements need to be satisfied if offsets are to meet their objective of no net loss. These relate to the **definition of offset requirements** (i.e. the extent and type of offset needed), and to **arrangements for implementing offsets and habitat banking** (i.e. the system that is put in place to ensure that offsets achieve their intended results).

In other words, achieving no net loss requires that:

- The extent and type of offsets required are sufficient to counterbalance the measured residual losses of biodiversity and ecosystem services; and
- The defined offset requirements are implemented effectively, with sufficient safeguards to ensure that their benefits are measured, verified and sustained over time.

Neither of these conditions is sufficient on its own to achieve no net loss. Even if offset requirements are adequately defined, they will not achieve no net loss unless properly implemented over the long term. On the other hand, even the best designed and delivered offsetting measures will not deliver no net loss if they are insufficient in extent to counterbalance ongoing losses of biodiversity and ecosystem services.

These two elements are important in the design of any offset or compensation scheme, and need to be considered when applying offsets under existing rules (such as any compensation for impacts under the Habitats and Environmental Liability Directives) as well as any new offset arrangements developed through the NNL initiative.

This study has provided an opportunity to improve our understanding of these two aspects of offsets design, and how they might work in the EU

The study focused on the two key design elements identified above, by examining:

- **The metrics used to determine offset requirements** for biodiversity and ecosystem services; and
- **The mechanisms used to secure long term conservation benefits**.

While the previous study by ICF GHK and BIO Intelligence Service explored these issues to some extent, drawing on EU and international experience, and guidance documents provided by BBOP and others, this latest study provided an opportunity to examine them in much more depth, and, importantly, to assess their applicability in the context of the EU and its Member States.

As well as providing an opportunity to assess international best practice in more detail, the study addressed issues such as:

- The different approaches to metrics and long term sustainability mechanisms adopted internationally;
- The lessons learned from existing experience of these issues in EU Member States and internationally;
- How metrics and long term conservation mechanisms can be applied to address current rates and patterns of loss of biodiversity and ecosystem services across the EU;
- How metrics and long term conservation mechanisms can be extended to cover ecosystem services as well as biodiversity, as required by the EU NNL initiative; and
- The requirements in applying these elements of offset design in the EU context, taking account of existing knowledge, capacity, policy frameworks, financial and regulatory systems.

This report therefore provides detailed evidence of these key aspects of offsets design, helping us to understand not only the elements to be taken into account for offsets to achieve no net loss, but also the practical issues and requirements in applying metrics and long term conservation mechanisms in different parts of the EU.

3 Metrics

3.1 Introduction

This review identifies and describes the range of metrics (which aim to measure losses and gains in biodiversity and ecosystems services resulting from residual project impacts and offsets) that are being used internationally for offsetting, including those that assess ecosystem condition and ecosystem services.

The review primarily draws on

1. Published papers and reports;
2. Reference to relevant websites, such as the BBOP website, which has an extensive on-line library of references and case-studies;
3. Offset scheme guidance and documentation;
4. Existing EU policy guidance of relevance to metrics. For example, the Habitats Directive includes an offsetting mechanism for Natura 2000 damage (in its Art. 6(4)) and Art. 12 species protection guidance); and
5. Targeted interviews with offsetting experts and scheme managers.

In particular the review focuses on the use of offsetting metrics in the following countries where offsetting is well established: Australia, France, Germany, South Africa and the United States or being trialled in Europe (the United Kingdom). Details of the offsetting and the use of metrics, including worked examples, in these countries are provided in Annex 1. Based on this information the review below firstly describes the aims of metrics, then outlines the main types of biodiversity and ecosystem service metrics and lessons that have been learnt from their use in the countries listed above. Drawing on this international experience, the review concludes with an assessment of the main advantages and disadvantages of each type of metric and a discussion of the key factors that need to be considered in designing a suitable metric for an offsetting scheme.

The implications of these findings are further discussed in the examination of options for offsetting metrics in section 5 and the study's conclusions in section 6.

3.2 Aims of metrics

3.2.1 Measuring losses and gains

Offsets aim to ensure no net loss (or a net gain) of biodiversity and this should apply to all components of biodiversity importance[2] that are significantly impacted. Therefore, to ensure this objective is achieved it is necessary to measure biodiversity losses from impacts (biodiversity debits) and the gains from offsets (biodiversity credits) in a practical and transparent way so that their equivalency can be compared.

Losses and gains should be calculated as follows:

- **Loss** = predicted situation for an affected area's biodiversity with no project impact minus the predicted situation for the affected area after avoidance, minimisation of impact and restoration.
- **Gain** = predicted situation for an offset area's biodiversity after the offset's conservation activities (e.g. restoration and/or management activities), adjusted for risk factors associated with these predictions, minus the predicted situation for the offset area with no offset intervention.

[2] For example, excluding invasive alien species and abundant pest species.

The calculation of biodiversity losses needs to take into account both:

- Areas that are directly affected by the activity of concern; and
- Areas that will not be completely converted or transformed by the activity but may be affected indirectly, resulting in a decline in conservation status, habitat quality/ integrity or status of key species populations. An example would be woodland whose condition is degraded as a result of more access following construction of a nearby road.

It is important to note that the reliable calculation of losses and gains using metrics is dependent on the accurate estimation of baselines. Baselines firstly need to be established in terms of the ecosystems, habitats, species and ecosystem services present at the impacted site, but also their wider overall local, regional, national and EU status. Such information forms the basis of the assessment of potential habitat values, which are explicitly taken into account in some metrics. Furthermore, as described in section 3.4, the assessment of baseline threats to habitats and species is necessary when setting outcome multipliers for risk aversion offsets.

Secondly, the calculation of impacts needs to take into account the baseline situation in terms of expected changes in habitats and species both at the impact site and the potential offset sites. Thus baselines are not an estimation of biodiversity at a particular point in time but a dynamic prediction of what is expected to occur on the basis of documented trends and the likely future impacts of political, policy, economic, social and other environmental developments.

There is normally significant uncertainty in the estimation of baselines and offset impacts, and therefore it is good practice to ensure that the calculation of losses and gains takes into account the precautionary principle, according to the CBD's definition, whereby "where there is a threat of significant reduction or loss of biological diversity, lack of full scientific certainty should not be used as a reason for postponing measures to avoid or minimize such a threat." The importance of this principle is recognised in the EU through its inclusion in Article 191 of the Treaty on the Functioning of the European Union, where it aims to ensure a high level of environmental protection through preventative decision-taking in the case of risk[3]. Application of the precautionary principle is particularly important in the case of risk aversion offsets because it is very difficult to estimate the added protection and management that these types of offset provide.

Measuring losses and gains in biodiversity and ecosystem services is not straightforward due to their complexity and context related variability. With respect to biodiversity, only in the very simplest of offset cases can losses and gains be measured directly in terms of numbers of individuals of particular species of conservation importance. More often it will be necessary to measure losses and gains in terms of species' habitat (e.g. the offset must provide at least as much gain of the species' habitat as was lost). However, habitats vary in their suitability for a species (i.e. in ecological terms, their carrying capacity) for example in relation to available food resources and breeding sites and the presence of competitors, parasites and predators. Spatial factors such as the size of the habitat patch and its functional connectivity to other habitat patches (e.g. to allow foraging, dispersal, colonisation and if necessary migration) are also of fundamental importance in terms of supporting viable populations or meta-populations[4] (Hanski, 1999; Levins, 1969; Opdam, 1991; Stavey *et al*, 1997). These habitat quality factors should therefore be taken into account when quantifying habitat availability for a species.

More often in Europe, offsets aim to address the general ecological values of habitats (i.e. their values in terms of communities of species or biotopes) rather than as habitats for individual species of conservation importance[5]. The general ecological importance and condition of habitats

[3] Common guidelines on the application of the precautionary principle are set out in COM (2000) 1 Final http://eur-lex.europa.eu/legal-content/EN/TXT/?qid=1417002187103&uri=URISERV:l32042

[4] i.e a group of spatially separated populations of the same species which interact at some level, and may therefore have a lower extinction risk than fully isolated populations.

[5] For this reason, the term habitats in this report normally refers to terrestrial or aquatic areas distinguished by geographic, abiotic and biotic features[5] (as defined in Article 1 of the Habitats Directive) and is therefore broadly analogous to the term biotope, which is often referred to in the German offsetting literature.

vary and measuring these proprieties is complex. Such issues make it difficult to simply quantify and compare expected biodiversity losses from developments with potential gains from offsets. Coupled to this is the complication of uncertainty concerning the likelihood that offset measures will improve habitat condition and increase the abundance of particular species.

To overcome such problems, metrics are used that define common currencies[6] and units of biodiversity so that the amount of biodiversity loss from impacts and the amount gained from offsets can be quantified and compared to establish if no net loss, or net gain are achieved (Quétier and Lavorel, 2011). If based on appropriate biodiversity components and attributes, metrics help ensure equity in the type, distribution and temporal delivery of biodiversity gains and the adjustment of offsets where necessary to guard against underperformance or failure (BBOP, 2012a; Gardner *et al*, 2013; Overton *et al*, 2012; Salzman and Ruhl, 2000a).

Metrics should as a minimum be applicable across sites, thereby allowing an impact in one location to be offset through actions elsewhere. If they are applicable across habitats then they can also allow offsets to involve different habitat types to those impacted. Furthermore, the creation of such a common currency not only supports individual exchanges of loss and gain, but establishes a standardised basis for quantifying losses and gains at a national or regional level, thus enabling habitat banking, which is essentially a mechanism for trading equivalent losses and gains (Eftec and IEEP, 2010).

The development of appropriate yet practical metrics is a major challenge as biodiversity is complex because it is multi-dimensional and scale-dependent and its value is highly context specific – thus all measures of it are proxies (Humphries *et al*, 1995). Ensuring appropriate biodiversity components are selected as the basis for metrics is therefore of fundamental importance as offsets will primarily deliver what is measured. In practice the most important biodiversity values can be assessed by measuring a few key properties either directly (e.g. counts of species of importance) or through surrogate indicators (e.g. assessment of habitat structural diversity and condition). However, there is currently no consensus on what biodiversity variables should be the basis for measure of biodiversity change, although attempts are being made to identify a set of Essential Biodiversity Variables (EBVs)(Pereira *et al*, 2013). However, these are of more relevance to high-level global monitoring needs (e.g. linking to the CBD Aichi targets) than assessing project-level losses and gains. Furthermore, due to the required scale of measurement, proposed EBVs tend to rely on remote sensing data, whilst in the EU more detailed and reliable data are normally available for assessing biodiversity and ecosystem service impacts from development projects (for example through existing species and habitat distribution maps and site-surveys required for Environmental Impact Assessments).

Primary biodiversity properties of impact sites and offset sites that are often considered and included in metrics, and are of most relevance to the EU, include:

- Their size.
- Their potential (i.e. inherent) relative biodiversity conservation value such as in terms of species-richness, distinctiveness, naturalness, biogeographical importance (which may be reflected in their listing in Annex 1 of the Habitats Directive or in national biodiversity action plans) or ecosystem service value (e.g. in terms of carbon storage or cultural importance), irrespective of its condition on the sites.
- Their importance for particular species of high conservation importance (such as those listed in Annex 1 of the Birds Directive and Annexes II and IV of the Habitats Directive or in national biodiversity action plans).
- Their actual relative condition (e.g. biophysical conditions, species and structural diversity, presence of keystone or functional species and integrity of ecological processes) and the viability of their species populations (e.g. chance extinction risks, genetic bottlenecks).

[6] In this context 'currencies' have nothing to do with placing a price on biodiversity components or ecosystem services, but instead measuring and comparing losses and gains in other ways

- Spatial factors, such as the distance between the impacted and offset sites, ecological connectivity to habitat networks, and the viability of wider meta-populations.
- Pressures affecting them, such as disturbance.

As described in BBOP guidance on no net loss and gain calculations (BBOP, 2012a), measures of such properties may result in metrics that quantify the suitability of a habitat for a single species. But more often biodiversity metrics are aggregated measures to provide, for instance, combined measures of loss and gain in habitat value. By contrast, as discussed further below, ecosystem services have distinct values that cannot be easily combined in a meaningful way, so their values are typically measured through individual metrics for each service.

After calculation of these primary biodiversity metrics, it is common practice to consider a range of secondary issues through multipliers and time discounting, which adjust the gains needed to accommodate potential uncertainties, time delays and social equity and distributional issues. Multipliers are discussed further in section 3.4.

3.2.2 Rules on what may be exchanged

By themselves metrics merely quantify amounts and do not fully indicate what exchanges of biodiversity may be appropriate to achieve no net loss. The issue of what is appropriate in terms of offsetting impacts is therefore a complex matter that should not be informed by the metric alone, but should be subject to **exchange rules**. Exchange rules consider the ‘kind’ of biodiversity being exchanged, and set out to control what kind of substitution is acceptable according to ecological principles and information on the status of biodiversity within the impacted area, but also in the wider context of regional, national and biogeographical considerations.

Three distinct issues are commonly considered in setting ‘exchange rules’:

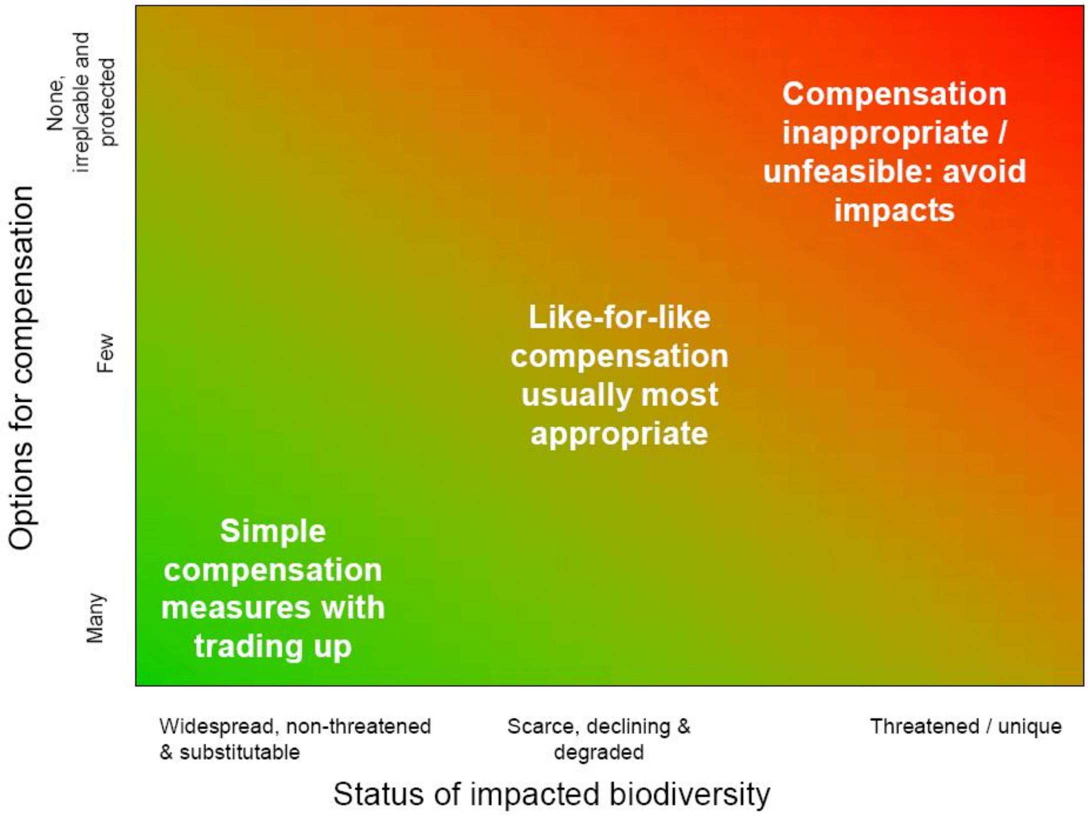
- How similar in terms of type and potential ecological value must the habitat gained through the offset be to habitat affected by the project’s impacts?
- Whether (and if so when, and to what extent) losses of habitat in very good condition can be offset by gains in habitat that start from a lower condition.
- The location of offsets in relation to the impacted area.

With respect to the first issue it is important to note that the need to provide like-for-like (also often referred to as in-kind) offsets generally increases as the potential ecological value of the habitat increases (Eftec and IEEP, 2010; Tucker *et al*, 2014, as illustrated in Figure 3.1). Thus, threatened habitats such as those listed in Annex I of the Habitats Directive should be subject to strict like-for-like compensation, as clearly indicated in Commission guidance (European Commission, 2007). Offsets for such habitats should therefore match those specifically defined in the Directives (e.g. *Tilio-Acerion* forests of slopes and screes), and not broader types of habitat (e.g. deciduous forest).

However, for habitats of moderate potential ecological value (and outside Natura 2000 and other prorated areas), it is appropriate to define habitats and therefore offset requirements more broadly. Indeed, it might sometimes be beneficial for overall biodiversity to offset them with other types of habitat where this is supported by a strong ecological rationale and robust evidence, especially if higher value habitat is the focus of the offset (i.e. trading-up occurs). Where habitats of low ecological value are impacted then it would often be appropriate to trade-up.

Importantly, it is generally considered to be inappropriate for offset habitats to be of lower potential ecological value than the impacted habitat, because it risks the loss of biodiversity attributes that are not fully captured in metrics. Thus, for example, although in England under the offsetting metric (see Annex 1.2), 6 ha of a low value habitat, such as an improved grassland is equivalent to 1 ha of a high value habitat, such as semi-natural forest, compensation through an offset that provides a larger area of the lower value grassland habitat would be unacceptable.

Figure 3.1 Appropriateness of compensation in relation to the importance of impacted biodiversity and availability of reliable compensation options



Source: adapted from BBOP (2009)

The second key exchange rule issue concerns the fact that habitats of equal potential ecological value can differ greatly in their actual value. Thus it is possible to offset losses by improving the condition of habitats, as well as protecting and restoring them. This might be particularly appropriate for habitats that are difficult to restore in reasonable time periods, such as forests. For example, forests in the UK are not rare or threatened but are widely affected by under-management, invasive alien species, and over-grazing by deer, and therefore there are numerous opportunities for enhancing their biodiversity by improving their condition. However, in most situations it would be inappropriate to consider offsets that aim to improve the condition of lower value habitats than those being impacted.

The third key issue often included in exchange rules relates to the location of the offset, and its service area (i.e. the area benefiting from it), in terms of the impacted site. In general offsets that are close to impacted sites are favoured because they are more likely to be ecologically similar and to be able to compensate the impacted beneficiaries of ecosystem services. However, as noted in the No Net Loss study (Tucker *et al*, 2014) requirements for local offsets can create practical supply problems where suitable sites are lacking, For instance, in France, requirements for offsets to be located close to the impacted site has resulted in a lack of suitable land becoming an issue. The need for more flexibility in allowing offsets to be delivered away from an impacted site is therefore being increasingly recognised, as for example shown through legislation changes in Germany, and the implementation of habitat banks and/or compensation pools (see Annex 1.4).

Most importantly, if the location of offsets is carried out through a strategic approach then biodiversity benefits may be greater than through following simple ‘local is best’ rules (Kiesecker *et al*, 2009; Kiesecker *et al*, 2010). Indeed, an important recommendation of the No Net Loss study was that biodiversity offsetting should be linked to landscape level planning, for example to support ecological networks or green infrastructure plans. However, when considering the location of biodiversity offsets it is important to ensure that local benefits and issues of social equity are not overlooked, as for example occurred in the USA, where wetland offsetting resulted in a redistribution of wetlands from urban to rural areas (Ruhl & Salzman, 2006). One way of dealing with this is to define 'composite offsets' with activities taking place in more than one location that in combination meet all offsetting requirements.

Such location issues are sometimes incorporated in metrics directly (such as in the wetland metric in France, see Annex 1.3) or as multipliers (e.g. in England, see Annex 1.2). However, they may also be dealt with through exchange rules, that for example require offsets to be within the same biogeographical area (such as in Germany, see Annex 1.4).

Clearly the assessment of whether no net loss, or other biodiversity objectives, is achieved by an offset is dependent on both metrics and exchange rules. Consequently, for the purposes of this study, the term 'metrics' has a broad meaning that encompasses both concepts (since both factors are essential for determining the nature, scale and measurement of offsets).

3.3 Types of metrics

The varied incorporation and treatment of biodiversity properties and ecosystem services gives rise to a large number of metrics. For example, there are more than 100 metrics used in the USA (BBOP 2009a,b) and over 40 in Germany (Bruns, 2007; Darbi & Tausch 2010; Busse 2013). Biodiversity metrics that are being used internationally can be classified in a number of ways, and a number of typologies of currencies can be found around the world. For example, BBOP (2012b), suggest that biodiversity currencies can be divided according to whether they are:

- Composed of direct or surrogate measures of biodiversity.
- Aggregated or disaggregated.
- Site-specific or context dependent.

However, this typology is not very relevant to current offsetting in Europe, where nearly all currencies are based on surrogate and aggregated measures of biodiversity and most are context dependent. Thus, the typology used for this study is a simple typology that relates to the biodiversity components that are the primary focus of the metric (i.e. habitats, species or ecosystem services) and the factors that are considered in assessing their ecological value. On this basis the following main types of metric can be identified (although there are many variations and overlaps):

- Habitat (biotope) area;
- Habitat (biotope) area x standard value;
- Habitat (biotope) area x site condition;
- Habitat (biotope) area x standard value x site condition;
- Species-focussed approaches;
- Habitat replacement costs;
- Ecosystem service specific metrics; and
- Economic valuation.

Some are used exclusively, and some in combination. Some are gaining and others losing traction according to a variety of factors such as the confidence of the conservation community that they serve as adequate proxies for biodiversity losses and gains, and also how straightforward, practical and cost effective they are. The use of these main types of metrics is further described in section 3.5.

In addition to these listed metrics, expert judgement may sometimes be used along with stakeholder discussions and negotiations. For example, 'qualitative verbal argumentative' methods are used occasionally in Germany (see Annex 1.4) and Quétier and Lavorel (2011) also noted that expertise may be used through 'Circumstantial reasoning' approaches to ensuring ecological equivalence of offsets. These subjective approaches may be used when the required information for the use of preferred metrics is unavailable or unreliable and it is felt that expert judgement is the best means of achieving a robust assessment. In fact given the complexities of biodiversity and often subtle differences in, for example, habitat types and condition, some degree of expert judgement is normally required and incorporated into metrics. Furthermore expert judgment is normally needed to interpret the metric values, for example in relation to their reliability, repeatability and local and regional context.

In practice, metric outputs may sometimes be used as an information source to inform negotiations between parties, rather than as a strict prescription for offset requirements (especially where offsetting requirements are not clearly backed up by legal obligations, such as in Sweden and England). Given the inevitable subjectivity of such an approach (and the fact that assessments by different professionals might vary) care should be taken to ensure consistency among such interpretations and the resulting decisions. It can help if these are properly recorded and explained (i.e. transparent).

3.4 Use of multipliers

It is common practice to adjust metrics using multipliers to address a number of issues that are not normally addressed within the metric. As discussed in a previous Commission habitat banking study (Eftec and IEEP, 2010) these typically include:

- Sources of **risk and uncertainty** that no net loss may be achieved (see Box 1).
- **Social equity and distributional issues**, which attempt to adjust for situations where offset gains are at least equal to losses, but at least some stakeholders do not get the same levels of gains from the offset (i.e. the losing stakeholders are not the gaining stakeholders), such as if they are in different locations. For example, stakeholders may be happy to walk twice as far to a park, which is a proposed offset for a lost park, if it is twice as large.
- **Desires to ensure a particular long-term outcome from averted risk offsets**, which takes into account existing conservation targets (which, for example, aim to protect a specified percentage of a threatened habitat) and/or possible constraints on 'endgame' protection levels when all of the targeted biodiversity resource is taken up either by development or offsets.
- **Temporary losses** of biodiversity and ecosystem service benefits for stakeholders.

Box 1 Sources of uncertainty in the assessment of biodiversity losses and gains

1. Biodiversity losses are not all accounted for in designing and implementing an offset. This may be because only a limited set of impacts is taken into consideration, or because only some biodiversity components have been considered.

2. Impacts on some components of biodiversity cannot be offset. In these cases, it is important to remove the uncertainty as to whether or not impacts may be non-offsetable (e.g. by undertaking additional in-depth biodiversity / ecological / social studies; assessing aspects of project design and predicted impacts, etc.) and undertaking relevant actions to respond to the findings.

3. Dissimilar biodiversity between impact and offset sites, which may be masked by the use of surrogate measures of biodiversity.

4. Uncertainty in offset performance due to a lack of data.

5. Uncertainty in the ecological system itself, including indirect impacts from secondary extinctions and due to the non-linear nature of biodiversity, ecological cascades, time-delayed ecological processes, natural disturbance regimes and stochastic ecological dynamics.

6. Uncertainty in offset implementation success, as a result of external factors (e.g. climate change, invasive species, fire and floods), technical issues concerning inadequately tested offset methods, financial failure and changes in political will.

Source: adapted from BBOP (2012a)[7]

As noted by BBOP (2012a), risk multipliers are grounded in the precautionary principle and serve to increase the basic size of an offset (as set by the underlying biodiversity currency and associated accounting model), thereby helping to account for concerns that the offset may not be sufficient to deliver a no net loss outcome. The calculation of appropriate risk multipliers should be based on empirical analysis (e.g. of offset failure rates, if they have been adequately monitored) or through consultations and negotiations with stakeholders with regard to distance and other equity issues. However, as Gardner *et al* (2013) point out, in practice multipliers are often generic rather than being linked to specific risks and mitigation measures (e.g. the probability that the desired specific vegetation type will establish and remain in the long-term). This is despite earlier research that indicated that for restoration offsets, the multipliers that are used are often too low (Moilanen *et al*., 2009). If calculated appropriately (i.e. probability of failing to achieve no net loss is minimized) very high multiplier ratios may be required (e.g. >1:100). The recently developed metric being used for pilot offsets in England did use the work by Moilanen *et al* as a basis for the calculation of risk multipliers for restoration/recreation offsets, some examples of which are shown in Table 3.1.

Table 3.1 Recreation / restoration risk multipliers and example habitats used in the Defra metric in England

Difficulty of recreation or restoration	Multiplier	Example habitats	
		Recreation	Restoration
Very high	10	Blanket bog, sand dunes, limestone pavement; all very high/impossible	None listed
High	3	Coastal vegetated shingle, mountain heaths and willow scrub	Limestone pavement, coastal vegetated shingle, wet heath
Medium	1.5	Coastal saltmarsh, lowland beech and yew woodland, lowland heathland	Coastal saltmarsh, lowland heathland, upland calcareous grassland
Low	1	Coastal and floodplain grazing marsh, hedgerows,	As listed for recreation and also lowland beech and yew woodland, lowland

[7] BBOP also list 'Time delays in offset delivery' but this is not included here as it is not regarded as a source of uncertainty in this report, but an issue of time preference.

Difficulty of recreation or restoration	Multiplier	Example habitats	
		Recreation	Restoration
		ponds, reedbeds	calcareous grassland, upland hay meadows

Source: (Defra and Natural England, 2012)

It should also be borne in mind that multipliers do not adequately address the risks of complete failure of a single offset as in such cases setting higher area requirements will not affect the outcome. The approach may, however, work collectively if other offsets address the same habitats and species. In most situations, a more appropriate risk management strategy recommended by Moilanen *et al* (2009) is to combine rigorous offset design with some form of hedge-betting, where a number of different offset solutions are carried out across a number of sites.

When averted risk offsets are used (i.e. areas of threatened habitat are given additional protection and management to secure their long-term conservation) then **multipliers may be used to secure desired large-scale or national conservation outcomes**. Such averted risk offsets tend to be used where restoration offsets are not feasible, because they cannot prevent an overall net loss of biodiversity, but instead aim to secure biodiversity that would otherwise be lost due to anticipated ongoing pressures. The amount of biodiversity that is secured over time is dependent on the impacted area to offset area ratio, such that a 1:1 ratio (one hectare of habitat lost for one hectare of habitat conserved) will, over time, lead to a 50% net loss of habitat. In this respect 'over time' means when all the available habitat is either developed or used for an offset and no more land is available for either activity. Thus, if a multiplier is applied to the ratio, then a greater area of habitat can be secured. For example, if a 2x area multiplier is applied then this would only lead to a 33% net loss of habitat over time.

Where a conservation target has been set for a biodiversity component (such as the retention of '30%') then the multiplier can be set to ensure that particular desired conservation outcome is achieved. As noted by Ekstrom *et al* (2008), this offsetting approach focusses on the 'endgame' and provides biodiversity benefits by setting a cap on the destruction of biodiversity that is higher than would otherwise occur.

Such risk aversion offsets are not widely used, and are not allowed in some countries (e.g. Germany), because they do not prevent net loss and have other drawbacks, such as the difficulty of predicting future losses and leakage effects (i.e. the displacement of a threat from one place to another, rather than its reduction), which makes offset gains and their additionality uncertain (Tucker *et al*, 2014). The use of endgame multipliers is also controversial because the developer is obliged to undertake an offset that is scaled according to earlier losses of the ecosystem that are not caused by them[8].

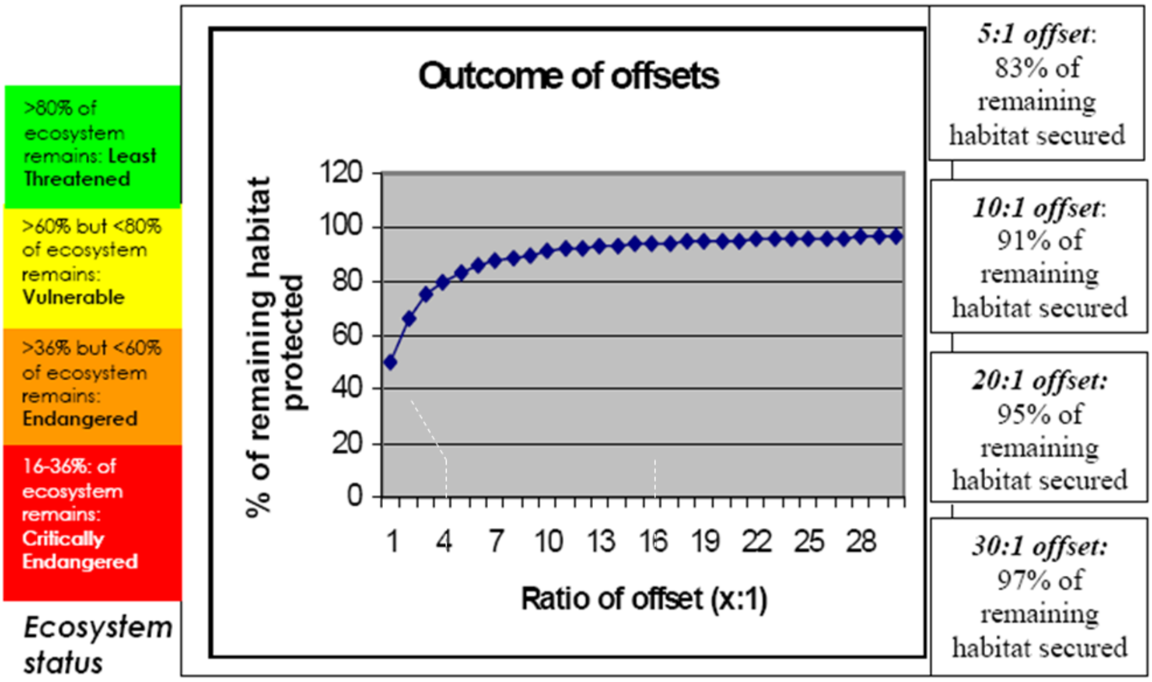
However, risk aversion offsets are used with 'endgame' multipliers under at least one current offset policy, the Western Cape of South Africa (see Annex 1.5). The rationale for this is that net loss is not realistic in South Africa, as it is a developing country, but there is scope for protection and management of many threatened habitats of high conservation importance Botha, 2009). Thus the focus is on ensuring offsets add priority habitats to the conservation they are protected and managed in the long-term. The ratio for offset requirements is primarily a multiplier that reflects the national conservation status of the affected ecosystem (i.e. it is based background rate of loss of the nation's ecosystems) and includes a built in 'safety margin'. Thus, in Table 3.2

, for example a 20 x multiplier is used for 'endangered' ecosystems, which would ensure that 95% of the remaining habitat would be secured if applied to all residual impacts from developments affecting the habitat.

[8] Kerry ten Kate, *pers comm*. 2014

According to Brownlie and Botha (2009), the system has a number of important advantages, including its simplicity and its explicit link between offset requirements and conservation priorities, which also result in a strong incentive for developers to avoid priority biodiversity areas. It does, however, require the regular updating of assessments of the conservation status of ecosystems in order to ensure that the endgame multipliers remain appropriate.

Table 3.2 The use of endgame multipliers in risk aversion offsets in the Western Cape province of South Africa



Source: Brownlie et al Western Cape Guidelines

Another use of multipliers is to adjust for **time preferences,** which attempt to take into account possible temporary losses of benefits and stakeholders normal preference for benefits sooner rather than later. Such multipliers are not normally applied to banking systems that are able to provide credits from offsets that are already in place. However, it might be appropriate to apply time preference multipliers to habitat banks if it is likely that their values change with time (e.g. if releasing credits for a 5 year old habitat scheme where it is demonstrated that the habitat is present, but has yet to deliver its functional value). An illustration of the use of multipliers for time preference is given in Box 2.

Box 2 Use of multipliers for time preference

Economists regularly use discount rates to express future flows of costs and benefits in equivalent units, in recognition that one euro is worth more today than it is in ten years' time.

Biodiversity offsets which involve the creation or restoration of habitats may take many years to deliver their intended benefits. In contrast, a development project may result in the immediate loss of biodiversity and ecosystem services. Assessment of losses and gains may therefore need to compare impacts that occur at different points in time. The principle of time preference tells us that a gain in biodiversity delivered by an offset in 20 years' time will not be sufficient to compensate for an immediate loss of equal magnitude. Discount rates can be applied to biodiversity offsets to take account of time preference, and to weigh up losses and gains achieved at different points in time.

Time preference multipliers have been included in the biodiversity metric being piloted in England. The metric uses the standard discount rate of 3.5% used by HM Treasury in the appraisal of public projects. Use of this metric assumes that 1 hectare of habitat today is the equivalent of 1.035 hectares of the same habitat in one year's time. Thus a multiplier of 1.035^n would be used to discount a gain in biodiversity achieved in n years' time. This means that an offsetting project that took 10 years to deliver its objectives would need to deliver 1.41 units of biodiversity gain for every 1 unit of immediate loss. Over 30 years a multiplier of 2.8 would be applied. When multiplied by other metrics for habitat condition, distinctiveness, risk and uncertainty, this can result in large cumulative multipliers being used to determine offset requirements. The application of these multipliers has a direct effect on the cost of delivering the offsets required.

It should also be noted that the use of a standard economic discount rate such as this reflects the value of biodiversity and ecosystems to people, rather than being based on ecological criteria. There is some discussion in the literature (including the TEEB reports) on the appropriate discount rates to apply to biodiversity impacts.

3.5 Use of the main types of biodiversity metric

3.5.1 Habitat area

In its most basic form, this metric is simply the area of habitat that is lost and gained. It is extremely simplistic as it assumes that all habitat types (although perhaps within a class of habitat), and indeed all hectares within a certain habitat type, are of equal value and condition.

This metric was used in some early offsetting schemes, such as those for wetlands in the USA (Salzman and Ruhl, 2000b). It is currently used in France in relation to wetland offsetting requirements defined in River Basin Management Plans (see Annex 1.3), despite the fact that wetland habitats are very widely defined, and as a result wetlands of high ecological quality can be offset with low quality habitats, leading to undetected biodiversity losses. Given its weaknesses it is not recommended for use in Europe except possibly for habitats of very low ecological value including for species (such as intensively managed arable land), where such habitats effectively form the lowest band of habitats covered by habitat area x standard value metrics. In such circumstances, as with other metrics, they would need to be combined with appropriate multipliers and exchange rules.

3.5.2 Habitat (biotope) area x standard values

A commonly used form of metric is based on standardised area ratios for individual habitats that reflect their different potential values. Ecological values typically reflect properties such as their naturalness, species richness and diversity, and rarity. Values may also take into account to some extent their potential to provide ecosystem services e.g. their sequestration and storage of carbon, water storage and purification, soil stabilisation, landscape aesthetic values and recreation (EASAC, 2009; Mace *et al*, 2011; Maes *et al*, 2011). This may be appropriate for some ecosystem services that are postively related to biodiversity levels (e.g. see Gamfeldt, *et al* 2013; Harrison *et al*, 2014), but others are not related and are best dealt with through ecosystem service specific indicators (as discussed below).

Habitats vary greatly in their potential values, and therefore the inclusion of ecological and ecosystem service values is probably the single most important element of all habitat focussed metrics beyond area. In this regard, it is important to point out that a habitat's potential value is not affected by its condition, which is area-specific and discussed in the next section. Its potential value might therefore also be regarded as its inherent value, irrespective of whether the area concerned reaches this potential. Thus, for example, a semi-natural habitat's inherent value is much greater than that of an artificial habitat.

The setting of ecological values for each habitat type can be based on empirical analysis of selected indicators (e.g. species richness or presence of threatened or endemic species). This provides a transparent, evidence-based objective means of setting the values. But such approaches are generally considered to be too narrow and selective, since data constraints may limit such assessments to better known biodiversity components (e.g. vertebrates and higher plants). Also approaches that set ecological values can become complex as, for example, species richness is affected by habitat area (Rosenzweig, 1995), and therefore this needs to be adjusted for. Therefore in practice inherent habitat values are often based on expert judgement (taking into account available information and general knowledge of habitats) and consultations. For example, such an approach was used to set distinctiveness bands for the metrics used for pilot offsets in England (see Annex 1.2).

Scientific evidence and stakeholder consultations, such as through national ecosystem assessments and the mapping of ecosystem services (such through the MAES initiative – see Box 7 below) should inform the valuation of habitats in terms of their provision of ecosystem services.

Area ratio metrics are widely used in Germany where many state or sub-state level metrics have been developed that use their own catalogue of standard biotope (i.e. habitat) types and corresponding value-based ratios (as described in Annex 1.4). These vary greatly in their sophistication and detail. For example, the Bavarian standard metric (described in Annex 1.4, Box 20) is very simple in only dividing habitats into the following three classes:

- **low significance (I)** - intensively used arable land and species-poor grassland, canalised water courses, other biodiversity-poor agricultural or amenity landscapes, etc.
- **medium significance (II)** - forest with non-local / non-native species, individual trees, tree groups or hedges without high biodiversity interest, extensively used grassland, floodplain habitats, etc.
- **high significance (III)** - mature semi-natural forest with a high proportion of locally appropriate tree species, mature species-rich hedgerows, copses and woodland edges, natural or near-natural freshwater landscapes, culturally significant landscapes etc.

Stated advantages of the Bavarian metric include its simplicity, which makes it easily understandable by non-experts. As a result offset assessments can be easily debated in the local political process, and it is argued that this transparency reinforces a broad appreciation of the cost-effectiveness of ecologically oriented planning that generates lower offset requirements (Busse *et al*, 2013). At the same time, the simplicity and flexibility of the metric leaves a relatively large room for discretion compared to other metrics, and therefore a high responsibility on planners and local authority to demonstrate its reasonableness and proportionality. It is also necessary for authorities to carry out additional qualitative analysis to adequately take into account specific local conditions. A problem with the metric is the low number of bands, which results in, for example, extensive grasslands that could include semi-natural meadows and pastures, being in the same category as plantations of non-native trees. Another problem is the relatively small difference in the ratios across the bands (i.e. from 0.2 for the lowest value habitats to 1 for the highest), which results in the size of the impacted area having a big influence on the offsetting requirement (Bruns, 2007). Thus large but low-value impacted habitats will require a relatively high amount of compensation in relation to their biodiversity value, whereas small-scale impacts on high-value habitats are probably often inadequately offset.

More sophisticated and ecologically realistic valuations have been developed in other regions by expert groups using various criteria, such as:

- The Baden-Württemberg list aims to characterise the ‘normal’ or ‘average’ form of that biotope in that region and is based on three criteria: naturalness, significance for threatened species and geographical and biological uniqueness (LfU Baden-Württemberg 2005b). Each of these criteria was scored for each biotope type between 1 and 5, and the combination located on an exponential scale from 1 up to the maximum biotope score of 64.

- The Nordrhein-Westfalen biotope type list values biotopes according to the average of their scores for naturalness, threat status/rarity, replaceability/restoration capacity and maturity (LANUV NRW 2008a).
- The Berlin method aims to characterise the basic 'optimal' value (which may be considered to be analogous to potential value) of each biotope type using the four criteria hemeroby, which is akin to naturalness (scored from 0 to 5), presence of threatened species (animals and plants) (scored from 0 to 7), rarity or threat status of biotope type (scored from 0 to 3), and diversity of animals and plants (scored from 0 to 5) (Land Berlin 2012). This biotope value is then differentiated according to two risk indicators on a scale from 0 to 20, and a connectivity score from 0 to 10 (which is a site-specific condition).

The use of an exponential scale in the allocation of basic biotope values in Baden-Württemberg is considered to be more appropriate, because it recognises the relative ecological value of the biotopes by ensuring that the higher scoring biotope types are progressively more valuable, and therefore more demanding to offset when damaged as well as more highly ranked as offsets (LfU Baden-Württemberg 2005b).

An example of a detailed ecological assessment based list of biotopes and their ecological values is provided in Table 3.3 below. This shows that in Nordrhein-Westfalen the assessment also takes into account condition to some extent, e.g. in relation to grassland species richness and the presence of typical trees in forests. Furthermore, some of the German metrics offer the option to upgrade or downgrade their relative values depending on site-specific factors in order to compensate for the simplicity of the standard valuation. For example, in Nordrhein-Westfalen, each biotope type score can be adjusted by up to 2 points up or down to account for local conditions e.g. lack of naturalness due to human impacts (LANUV NRW 2008a). Thus, in practice, when standardised value based metrics become detailed they overlap to some extent with the site condition based metrics discussed below.

Table 3.4 provides an example of how a habitat area ratio metric based on standard potential habitat values is used to measure impacts and calculate offset requirements (see Annex 1.4, Box 21 for further details of the method). The example compares habitat losses and gains in eight subareas of the impact area.

Table 3.3 Derivation of biotope type values in Nordrhein-Westfalen

Explanation: Each of the biotope types in Nordrhein-Westfalen is scored according to naturalness, replaceability, maturity, and threat status or rarity. The four qualifying factors are equally weighted and the overall biotope value is derived from the numerical average. Source: LANUV NRW (2008)

Biotope Type	Description	Naturalness	Replaceability	Maturity	Threat Status/Rarity	Overall Value	Modifying Factor	Other Factors
HA0, aci	intensively cultivated arable, poor to absent wild plant flora	2	1	2	1	**2**	-1 if highly intensive cultivation without any wild plant flora	
EA.xd1.veg1	species-rich meadow, medium to poor quality (3-4 indicator species present)	5	3	5	6	**5**		certain types are legally protected biotopes; certain types are EU-protected Annex I habitat 6510 lowland- or 6520 upland-hay meadow; consider list of species associated with meadows
AA.70.ta5.m	forest (10 types[9]) with 50 < 70% habitat-typical tree species, tree size – young (diameter at 1.30 m less than 13 cm), habitat structure medium to poor	5	3	4	4	**4**		certain types are legally protected biotopes; consider list of species associated with forest
RO.wf6	river - slightly modified	5	8	4	4	**5**	some biotopes are considered to be irreplaceable	certain types are EU-protected Annex I habitat 3260 or 3270 ; consider list of species associated with flowing water

[9] These are listed in the overall biotope catalogue for Nordrhein-Westfalen, ie AA = , AB == , AC = , AD = , AE = , AG = , AM = , AP = , AQ = , AR =)

Table 3.4 Example of biotope valuation using the Sachsen method (part A) (SMUL 2009b)

Subarea	Negatively affected biotopes			Development			Net score	Area (ha)	Net value loss
	Biotope Code	Biotope type	Score	Biotope Code	Biotope type	Score			
1	7 - WLE	Oak – hornbeam woodland	27	9 5 100	Road (sealed)	0	-27	1.35	-36.45
				9 5 600	Grass verge	5	-22	0.8	-17.6
									∑54.05
2	4 -GFY	Other damp grassland, species rich (presence of rare species such as *Orchis mascula*)	25	9 5 100	Road (sealed)	0	-25	0.3	-7.5
				9 5 600	Grass verge	5	-20	0.15	-3
									∑10.5
3	*03220*	Stream with straightened channel / artificial banks and semi-natural elements	18	9 5 100	Road (sealed)	0	-18	0.025	-0.45
				9 5 600	Grass verge	5	-13	0.01	-0.13
									∑0.58
4	4 - GFY	Other damp grassland, species rich (presence of rare species such as *Orchis mascula*)	25	9 5 100	Road (sealed)	0	-25	0.25	-6.25
				9 5 600	Grass verge	5	-20	0.1	-2
									∑8.25
5	*06320*	Intensively used permanent mesic grassland	10	9 5 100	Road (sealed)	0	-10	0.8	-8
				9 5 600	Grass verge	5	-5	0.3	-1.5
									∑9.5
6	6 5 100	Hedge (more than 60 years old)	25	9 5 100	Road (sealed)	0	-25	0.01	-0.25
				9 5 600	Grass verge	5	-20	0.003	-0.06
									∑0.31
7	*10120*	Intensively used arable field	5	9 5 100	Road (sealed)	0	-5	1.25	-6.25
				9 5 600	Grass verge	5	0	0.4	0
									∑6.25
8	*06320*	Intensively used permanent mesic grassland	10	9 5 100	Road (sealed)	0	-10	0.8	-8
				9 5 600	Grass verge	5	-5	0.55	-2.75
									∑10.75
									∑54.95

3.5.3 Habitat (biotope) area x site condition

This type of widely used metric is based on a multiplication of the area of the impacted habitat by the change in ecological condition (e.g. a change in the percentage of its potential condition) resulting in a currency that is often referred to as ‘habitat hectares’. This refers to the Habitat Hectares metric developed by (Parkes *et al*, 2003) that is used in the Victoria State offsetting scheme in Australia (as described in Annex 1.1 and summarised in Box 3).

Box 3 Summary of the habitat hectares metric used in Victoria, Australia

The habitat hectares metric provides a way of calculating losses and gains in vegetation condition for each distinct Ecological Vegetation Class (EVC) in Victoria based on units of measurement that take into account the area affected and the quality or condition of the vegetation impacted in relation to benchmarks for 10 habitat attributes for each EVC, such as: number of large trees, canopy cover, number of understorey lifeforms, cover of weeds, recruitment, cover of organic litter, abundance of logs, patch size, proximity of remnant vegetation and distance to core area. The attributes in the benchmark are weighted according to their significance to the overall condition of the system. A user measures each attribute at the impact site before the impact and the predicted score after the impact, comparing the measurements against the benchmark which represents the pristine condition of the habitat in question. The scores for each attribute are then added (according to their weightings) to provide an estimate of the site's condition expressed as a percentage pristine condition. The area of the habitat is then multiplied by this percentage change in condition. The same approach is used to estimate the gains at the potential impact sites, comparing the actual measurements before the offset activities start with predicted realistic outcomes from the offset, again compared with the benchmark levels.

Put most simply, the loss of 100ha of forest at '50% quality' is expressed as the metric of 50 "habitat hectares" and can be compensated for with offset gains of 50 habitat hectares. This can be achieved, for example, through the gain of 25% of 'condition' (=quality) over an area of 200ha, or 100% 'condition' over an area of 50ha. Offsets are, however, also covered by exchange rules which for example ensure that gains in low condition areas cannot be substituted for losses of high condition areas.

The metric has been revised, but the key principles remain the same[10]

It is relatively sophisticated metric compared with area x standardised value metrics because it takes into account the actual condition of the habitat rather than being based on average or theoretical potential ecological values as under standardised value based metrics (discussed above). Thus it enables the use of offsets that improve condition (i.e. through enhancement and restoration) as well as through re-creation. However, because the metric does not incorporate habitat values and the condition assessment is specific to each habitat type, it can only be used for like for like offsets. But this may not be a problem if it is only used for habitats of high ecological value as strict like for like offsetting is appropriate for such habitats.

The rationale for the metric is that habitats of the same type can vary considerably in their ecological condition, particularly semi-natural and natural habitats. For example areas of the same habitat may vary in terms of their:

- naturalness of biophysical conditions (e.g. soil type, structure, hydrology);
- integrity of ecological processes (e.g. erosion, succession, fire regimes, nutrient cycling);
- proportion of species that are native and characteristic of the habitat type;
- structural naturalness (e.g. in terms of vegetation layers);
- age (e.g. presence of veteran trees and deadwood);
- integrity of species communities, food webs and presence of keystone species (e.g. higher predators);
- habitat patch size; and
- ecological connectivity to other areas of the same or similar habitat.

It is, therefore, theoretically possible to quantify habitat condition according to such key attributes in comparison to some form of benchmark or reference state based on ecological principles and empirical evidence (Gardner *et al*, 2013). This accords with the concept of favourable conservation status, as defined in Article 1 of the Habitats Directive (see Box 4), which the Directive aims to maintain or restore[11]. In simple terms, favourable conservation status can be described as "*a situation where a habitat type or species is prospering (in both quality and extent/population) and with good prospects to do so in the future as well"* (European Commission, 2011).

[10] For details see http://www.depi.vic.gov.au/environment-and-wildlife/biodiversity/native-vegetation/native-vegetation-permitted-clearing-regulations/assessing-permits-to-remove-vegetation

[11] Article 2(2) of the Habitats Directive states that "*Measures taken pursuant to this Directive shall be designed to maintain or restore, at favourable conservation status, natural habitats and species of wild fauna and flora of Community interest*."

Box 4 Definition of favourable conservation status for habitats and species under the Habitats Directive

Under Article 1(e), the conservation status of a natural habitat will be taken as 'favourable' when

- its natural *range* and *areas* it covers within that range are stable or increasing, and
- the specific *structure and functions* which are necessary for its long-term maintenance exist and are likely to continue to exist for the foreseeable future, and
- the conservation status of its *typical species* is favourable as defined in (i).

Under Article 1(i), the conservation status of a species will be taken as 'favourable' when

- *population dynamics* data on the species concerned indicate that it is maintaining itself on a long-term basis as a viable component of its natural habitats, and
- the natural *range* of the species is neither being reduced nor is likely to be reduced in the foreseeable future, and
- there is and will probably continue to be, a *sufficiently large habitat* to maintain its population on a long-term basis.

Source: Council Directive 92/43/1992 (Emphasis added)

In practice the quantification of condition is complex because numerous habitat attributes are of importance and problems arise when measures of these are combined (e.g. in terms of ensuring appropriate weightings and avoiding double-counting or masking of biodiversity components). Habitats also vary considerably between areas according to local conditions and their position in successional pathways, which makes it difficult to identify appropriate benchmarks that are generally applicable to the habitat. Furthermore, good examples of habitats that can act as benchmarks may no longer remain, especially for rare and threatened habitats. Consequently, as discussed more fully in Annex 1.1, the habitat hectares system developed by Parkes *et al*, (2003) has been criticised, notably by McCarthy *et al* (2004) in relation to the use of habitat attribute benchmarks that represent stable climax vegetation communities (rather than dynamic communities that responding to recurrent disturbances) the strong influence of assessor variability, the rationale for simply multiplying the habitat area by the quality score, (as condition will be affected by the habitat area), and the adding of attribute values which implies that the attributes are substitutable.

The assessment of good condition also becomes particularly problematical in terms of human modified habitats (which predominate in the EU). In fact the assessment of the condition of highly artificial habitats, such as intensively used grasslands and crops, may be inappropriate (e.g. see Defra metric for England in Annex 1.2). Further discussion of the problems of assessing condition in relation to appropriate benchmarks is provided in BBOP's guidance on the assessment of losses and gains (BBOP, 2012a).

It is also worth noting that the metric is much more demanding in terms of data requirements than standardised habitat value based metrics, and normally requires field assessments of habitat condition over the impacted area and at the offset site. It is therefore likely to be a relatively costly metric, although no information was found in the literature to quantify this.

Despite some of the criticisms and limitations outlined above, the habitat hectares approach is widely considered to be a more rigorous than the use of area x standard habitat value metrics, whilst being pragmatic and applicable to a broad range of biodiversity components. It therefore continues to be used in Victoria[12] (although it has been recently revised) and has been widely adopted and adapted internationally, for example in Western Australia (Hajkowicz *et al.*, 2009), South Africa (Kotze, 2005) and at a suite of projects worldwide in accordance with BBOP guidance. Some adaptations have attempted to deal with some of the problems outlined above, whilst others have incorporated other factors into the system.

Although the habitat hectares approach could in theory be used to assess the adequacy of compensation measures required under the Habitats Directive, it does not appear to have been widely used for this purpose, if at all. This may be appropriate, because given the importance of such habitats and the difficulty of quantifying condition in practice, compensation through improvements in condition may not be sufficiently reliable. Furthermore, given that the maintenance and restoration of favourable conservation status is an obligation for Member States under the Habitats Directive, offset gains through condition improvements would need to go beyond these obligations to ensure additionality.

3.5.4 Habitat (biotope) area x standard value x site condition

These metrics combine consideration of standardised potential habitat values (expressed as habitat area ratio requirements) with relative site condition assessments to provide an aggregated metric. For example, the majority of assessments in offsetting schemes in the USA make use of an area measurement and a value multiplier, and some incorporate an approximate quality assessment based on expert opinion (Briggs *et al*, 2009).

In the EU a metric of this type was developed recently for use in pilot offsets in England (DEFRA and Natural England, 2012). This is described in detail in Annex 1.2, but in essence it develops the habitat hectares approach to produce a metric that is based on the multiplication of a standard national ecological value score for each habitat type by a site habitat condition score. Habitat values are referred to in terms of distinctiveness, which fall into three bands based on parameters such as species richness, diversity, rarity (at local, regional, national and international scales) and the degree to which a habitat supports species rarely found in other habitats. Habitats listed in the Biodiversity Action Plan are in the top band and are allocated a score of 6. Other semi-natural habitats are in the second band and score 4. Artificial habitats, such as intensive farmland, are in the lowest band and score 2. It is noteworthy that the increased weighting for natural and semi-natural habitats compared to artificial habitats is relatively modest.

The condition of sites is assessed according to three bands, using a relatively simple condition assessment methodology. The condition bands are scored 1, 2 and 3, which when multiplied by the habitat distinctiveness scores results in product scores for each combination that range from 2 to 18.

The key advantage of the metric is that it facilitates both offsets that aim to increase the value (distinctiveness) of habitats e.g. by converting a low value habitat into a high value habitat (as do standard value metrics) and offsets that aim to improve condition (as do condition metrics) and combinations of these. However, exchange rules generally require offsets to be like-for-like in the top band, or at least within the same band; within the same band or through trading up for the second band, and through trading up when habitats in the third band are impacted.

Overall it seems that the use of this type of metric has been generally accepted by stakeholders in England, and Tyldesley *et al*. (2012) considered that the metric is valid and works, but noted that it does not assess impacted habitats in the context of their wider ecological setting. According to Defra, due to its simplicity it only takes about 20 minutes to apply. Whilst this may be an underestimate, and the time required will vary according to site complexity and size, it suggests that the cost of applying the metric should be relatively low. However, the actual metric is also widely considered to be overly simplistic and as a result the Environmental Audit Committee (2013) recommended that “If biodiversity offsetting is introduced, its metric for

[12] For details see http://www.dse.vic.gov.au/conservation-and-environment/native-vegetation-groups-for-victoria/vegetation-quality-assessment-manual

calculating environmental losses and gains must reflect the full complexity of habitats, including particular species, local habitat significance, ecosystem services provided and 'ecosystem network' connectivity".

3.5.5 Species-focussed approaches

The rationale for species focussed approaches

Most offset metric schemes are currently based on habitat assessments as these are thought to be more practical and provide balanced assessments of biodiversity importance and condition. However, especially where highly threatened and/or protected species are involved (such as those covered by Article 12 and listed in Annex 4 of the Habitats Directive), some metrics focus on the expected impacts on defined species' populations. There are a number of reasons for this, some of which were identified in a recent workshop on offsetting metrics in England (Howard *et al*, 2013) and are described in Annex 1.2. In summary, metrics that assess general habitat values and condition may not provide a reliable basis for safeguarding individual species of high conservation importance because habitat metrics do not normally consider the following issues:

- Habitat structure (the arrangement of features such as soil layers and types of plants) affects the occurrence of species.
- Some species depend on a mix of different land cover types, vegetation types and landforms.
- The management of a habitat affects the mix of species present and their abundance.
- The existence of predators, pests and diseases affects the distribution of individual species.
- The presence or absence of a species at a particular site is often determined by the interaction of a number of factors.

It should also be borne in mind that highly threatened protected species are likely to have narrow and specific ecological requirements, as this is often a major underlying reason for their threatened status. Therefore it is especially important that metrics are sufficiently refined to be able to distinguish the suitability of habitats for such species.

A drawback of the use of species-focussed approaches is that in many situations more than one species merits assessment. As described below, species metrics are normally complex and require a considerable amount of data and therefore the use of metrics for more than a few species is normally impractical. The interpretation and communication of the results of metrics for several species also becomes complex. If the number of assessed species is low then a logical approach is to define offsetting requirements according to the most sensitive species' needs. Then, if all the species have similar habitat requirements, it may be assumed that offsetting will ensure no net loss will occur for all of them.

An alternative approach is to use a species metric for one selected umbrella species (i.e. an indicator species representative of the set of species of high conservation importance). The umbrella species for a habitat is that which is considered to be most sensitive to the impacts of the development, and will therefore require the largest / highest quality offset. However, this requires good knowledge of the various species that may be impacted and high confidence that the umbrella species is indeed representative of the requirements of the other species and the most sensitive and in turn most demanding in terms of offset requirements. It has not been possible within the scope of this study to examine whether these assumptions are in fact reasonable and borne out in practice.

Habitat Evaluation Procedures (HEP)

In practice losses and gains in species are normally measured in terms of the species' habitat and are assessed through procedures such as the Habitat Evaluation Procedures (HEP). These were initially developed by the US Fish and Wildlife Service (USFWS) in 1976[13]. The rationale for

[13] http://www.fws.gov/policy/esm102.pdf

the HEP is that areas affected by projects and those identified for offset activities can contain various habitats, and that these habitats can have different suitabilities for species that may occur in that area that can be quantified through habitat suitability models, resulting in a Habitat Suitability Index (HSI). Provided that the extent of the different habitats can be measured, the overall suitability of an area for a species can be represented as a product of the area of each habitat and the HSI index for each habitat for the species, which is referred to as Habitat Units (HUs).

The calculation of HEPs and application to offsetting a project involves a number of steps, which, according to Treweek (1999) can be summarised as follows:

- Selection of key wildlife indicator species.
- Review of habitat requirements for selected wildlife indicator species.
- Definition of study limits.
- Identification of plant community types and associated landscape units needed to estimate habitat suitability.
- Collection and field measurement of a range of significant variables by plant community.
- Development of Habitat Suitability Index (HSI) models.
- Determination of HSIs (inadequate to optimum) for wildlife evaluation species by plant community type.
- Determination of habitat supply, i.e. habitat units (HU), a product of HSIs and habitat area.
- Description of baseline habitat conditions in terms of HUs.
- Projection of future habitat conditions, with anticipated project impact and without (i.e. baseline)
- Quantify the difference in HUs.

The HSI is based on the assumption that habitat influences animal distribution and that certain habitat variables can be measured (e.g. vegetation composition) that are strongly correlated with the ability of an area to support a given species (i.e. its carrying capacity). A numerical system of 0 to 1 is used to index the suitability of habitats for each of the selected species. A score of 1 would be given if the habitat provides for all a species' needs or provides an integral part of its lifecycle without which it would not be able to maintain its existence. On the other hand, if a habitat provides low abundance of food, no shelter and therefore is of poor quality a low score is applied. Habitats not providing any support to the species at all would have a value of 0.

As pointed out by Treweek (1999) the reliability of HEP and HUs is therefore greatly dependent on the ability of the user to assign a well-defined and accurate HSI to the selected evaluation species, and more specifically, to identify clear relationships between carrying capacity and the modification of the specific environmental variables. The selection of evaluation species also has an important influence on the outcome.

Clearly this is a complex process that is dependent on good knowledge of the ecological requirements of a species (and their variation) and the ability to map out the different habitat types and reliably predict the likely impacts of the development on the habitat area and properties of important to the species. The HEP metric therefore takes time to calculate and is likely to be costly in most cases. For such reasons it does not appear to have been taken up widely outside the USA, where it is widely used by land managers (see Annex 1.6). However, an adapted version of the process is being successfully used in Somerset, UK. This is described in Annex 1.2.

A more specialised metric, Landscape Equivalency Analysis (LEA), focusses on quantifying habitat loss and fragmentation in relation to the conservation of endangered species covered by the US Endangered Species Act (Bruggeman 2005, 2009). This develops Habitat Equivalency

Analysis (described further in section 3.5.7) to provide a generalised landscape-scale accounting system that assigns conservation value to habitat patches based on patch contributions to abundance and genetic variation at the landscape scale. The abundance and genetic variation of the species in question is treated as an ecological service in LEA, with the goal being to identify specific landscape configurations that would maintain equivalent level of these ecological services despite changes in landscape structure arising from activities such as offsetting. The level of service is summarised per unit of resource (e.g. per hectare) and the quality of the two sites is based on discounted Service Acre Years. These are a time-weighted measure of resource quality based on an area-weighted measure of service flows from the resource. To test this, the LEA method uses meta-population genetic theory to estimate sustainability criteria against which all trades are judged.

Clearly this is a particularly sophisticated and ecologically rigorous approach, and is therefore probably too demanding in terms of data requirements to be practical for general offsetting. Such approaches may, however, be more appropriate for the assessment of required compensatory measures for threatened protected species, such as those listed in Annexes 2 and 4 of the Habitats Directive.

3.5.6 Replacement costs

This type of metric is occasionally used in Germany (Darbi and Tausch, 2010), and is simply the average cost of replacing the lost habitat (including also planning and maintenance costs) multiplied by its area. The offset must then create an area of habitat of equivalent cost. For example, in the city of Berlin the method is designed for small-scale low biodiversity-value impacts associated with inner city developments, where commonly used offset measures are tree and shrub planting, green roofs, semi-permeable surfaces and other sustainable urban drainage measures. Specific protected species issues must be considered separately (e.g. bat habitat protection).

This metric is appropriate for calculating required payments for fee-in-lieu based offsets or the related trust administered conservation credits that are policy options suggested in the no net loss policy options study (Tucker *et al*, 2014). However, it is important to note that except for the simplest and lowest biodiversity value habitats it needs to be informed by other metrics in order to specify, as a minimum, what habitat type needs to be replaced. Ideally habitat condition should also be taken into account.

3.5.7 Ecosystem service metrics

Introduction

In addition to its global 'existence value' to humanity, biodiversity plays an important role in the regulation of ecosystem processes and the provision of ecosystem services such as food, genetic resources, medicines and tourism (Mace *et al*. 2012).

Values associated with biodiversity conservation are usually regarded as distinct from ecosystem service values, not least because species of high conservation value may not correspond spatially or temporally with ecosystem services (Naidoo *et al*. 2006). Because values associated with ecosystem services vary significantly from one site to another, establishing 'like-for-like' compensation for biodiversity offsetting is fraught with difficulty. Complex trade-offs are necessary where enhancing one service results in depletion of another. Local communities may be unwilling to accept offsets for biodiversity occurring away from the impact site if this entails the loss of locally valued ecosystem services.

The Convention on Biological Diversity's Aichi targets (CBD, 2010), the International Finance Corporation's (IFC) Performance Standard 6 and the Biodiversity Strategy of the European Union all explicitly include ecosystem services within their 'no net loss' goals or requirements (IUCN, 2013), whilst the most recent EC Working Group on NNL relates to 'ecosystems and their services'.

The implications of this in the context of offsetting are that (i) any compensation or offset should ideally deliver both biodiversity and a range of ecosystem services (e.g. carbon capture, water remediation, flood risk mitigation, etc.); (ii) the net effect of project impacts and offsets should be considered at the community level, with explicit consideration of the livelihood and cultural values of biodiversity.

An effective offsetting system therefore needs to ensure- as far as possible- that:

1. Local people do not experience a loss in their livelihood, amenity and cultural values associated with biodiversity as a result of a project, or indeed its offset; and
2. Compensation of biodiversity (from a perspective of NNL) goes hand-in-hand with compensation for any loss of associated ecosystem service benefits. Clearly, displacing all ecosystem services occurring in urban areas to the countryside is not socially or environmentally desirable, and runs counter to the Green Infrastructure agenda. Better utilisation and development of data infrastructure relating to ecosystem services and biodiversity can assist in this regard.

Local stakeholder engagement can clearly therefore play an important part in developing an understanding of the impacts and net effect of offsets on local values. IFC Performance Standard 6 emphasises the need to engage and consult local stakeholders as part of the process of offset development and Critical Habitat identification. Whilst such processes can be highly beneficial for ensuring the robust design and acceptance of offsets, it is however important to consider that the original stakeholder engagement processes necessary in developing countries may represent a duplication of previous consultations in an EU context, such as for the establishment of Natura 2000 sites.

EU Biodiversity Strategy and NNL of Ecosystem Services

The headline target of the EU Biodiversity Strategy is to *halt the loss of biodiversity and ecosystem services by 2020*, while the No Net Loss Initiative refers to NNL *of ecosystems and their services.* It follows that offsets – as measures to compensate for residual losses – must be capable of compensating for losses of both biodiversity and ecosystem services.

This has important implications for metrics, since, to demonstrate no net loss through offsets, the measurement of gains and losses should in some way take account of changes in ecosystem services.

Issues and Challenges

While the specification and application of accurate and workable metrics for biodiversity presents significant scientific and practical challenges, as described above, the challenges of measuring losses and gains in ecosystem services are greater still. These challenges relate to:

- **The fact that the benefits provided by most ecosystem services are context specific.** With the possible exception of carbon storage and sequestration (the benefit of which contribute to global climate objectives) the benefit provided by an ecosystem service in any location will depend on the demands for the service, the scale and value of which is shaped by local circumstances. For example, flood alleviation through water storage in the upper catchment is not important if a river is not prone to flooding and/or flooding is not a problem (e.g. because there no settlements or sensitive valuable land uses present in the floodplain). Thus ecosystem service measurements need to take into account the local context specific importance of each potential service.
- **The number of ecosystem services that need to be measured.** This depends on the classification system used, as well as the local significance of these services, but the Millennium Ecosystem Assessment, for example, identifies 21 types of ecosystem services across the four groups of provisioning, regulating, cultural and supporting services, while the CICES (Common International Classification of Ecosystem Goods and Services) identifies 9 classes and 23 different groups of ecosystem services under three themes (provisioning, regulation and maintenance, cultural). IFC Performance Standard 6 suggests that an initial

scoping exercise is carried out in order to determine 'priority' ecosystem services (in terms of impacted communities and project dependencies) for any given case;

- **Challenges in quantifying service delivery**. While some services, such as food production, carbon sequestration and recreation may be relatively easily measured (even if data are not always available), others such as flood risk management, water purification and erosion control may be highly locally specific and difficult to measure without in depth studies;
- **Challenges in combining data on different ecosystem services**. If metrics are to assess changes in ecosystem services as a whole, they need to somehow combine information on different services. This presents substantial challenges, since, even if these individual services can be quantified, they are likely to have different significance and value, and be measured in different units. A second major challenge with combining data on different ecosystem services is that gains in one ecosystem service (e.g. carbon sequestration) can mean a loss in another (e.g. food provision), so great care needs to be taken when reporting outcomes for several ecosystem services that they do not mask net losses for individual ecosystem services.

As a result, there is very little evidence or experience internationally in relation to metrics for offsetting losses of ecosystem services. Indeed, even in the UK where knowledge of ecosystem services is relatively well developed, a seminar organised by the Natural Capital Initiative in 2010 concluded that “the data which exist in the UK are not sufficient to allow offsetting for ecosystem services. Data collection must be augmented to encompass ecosystem services, and existing data brought together”.

A key issue in applying offsets to ecosystem services is the question of what is meant by no net loss of ecosystem services. Given the number of different ecosystem services, and the trade-offs between them, achieving no net loss of each and every individual service is unlikely to be feasible. An approach which required offsets for combined losses in ecosystem services would be more feasible, allowing for gains in some services to compensate for losses in others. However, this would raise challenges for measurement and metrics design.

Possible Options

Given these challenges, some options for incorporating ecosystem services into a system of metrics can be identified. These may include:

- **Establishing separate metrics for biodiversity and ecosystem services.** There are challenges in combining different ecosystem services, which result in risks that gains in one ecosystem service (e.g. carbon sequestration) can mean a loss in another (e.g. groundwater recharge), and poor data on ecosystem services mean that these net losses may be magnified or unnoticed within the metrics. Further, seeking to balance net ecosystem service benefits will not necessarily have a positive effect on biodiversity more generally. It is useful therefore to consider the two elements (biodiversity and ecosystem services) independently. This could be achieved by calculating losses and gains of biodiversity as a first step, and then assessing changes in ecosystem services as a separate, additional exercise. The metrics used would have to ensure no net loss or net gain in both.
- **Use of proxy metrics**. In practice, any metric will need to simplify the change we are concerned with in order to be workable. Metrics used for biodiversity do not measure all changes in species, habitats and genetic diversity, but use reasonable proxies such as the area, condition and distinctiveness of ecosystems affected, and attributes could be included that code for ecological function and process. These may provide a reasonably good proxy for overall ecological functions and processes and thus for the delivery of ecosystem services, to which more specific conditions can be applied as necessary at the site level.
- **Quantitative metrics for single ecosystem services**. While it may be difficult to define combined metrics for ecosystem services, metrics for single services are more feasible. For example, there is widespread use of metrics for carbon storage and sequestration, and metrics are being developed for a range of other services (e.g. pollination), and metrics could

also be applied to other services (e.g. provision of recreational green space). Therefore in cases where a particular ecosystem service is significant and measurable, it should be possible to define useable metrics to complement the main metrics to be used for biodiversity offsets.

- **Use of semi-quantitative / expert judgement based metrics**. Given the complexity of ecosystem services and problems with measuring them in practice, metrics often use a combination of quantitative measurements and judgement. For example, in Germany metrics have been used to define NNL of ecosystem functions such as those provided by soil (see section A1.4.2).
- **Setting the exchange rules** to cover a 'like for like' approach for ecosystem services (separately to like-for-like exchange rules set for biodiversity). As discussed in section 3.2.1, attaining NNL entails exchange rules as well as metrics. Exchange rules ensure the kind of biodiversity gained is similar to that lost ('like for like or better') and the metrics ensure that the amount of gain balances or exceeds the loss. There may be challenges (as described in this section, some of which can be addressed) with metrics for ecosystem services, but it should be possible for exchange rules to require inclusion of qualitative activities to restore or conserve ecosystem services. For instance, there could be a requirement for an assessment of the types and significance of the services affected and that the offset will deliver similar types and levels of ecosystem service. Such a test should help to ensure that damage to a site that provides an important flood management service could not be compensated for at a site that offers no such service, or that development of an important recreational area close to an urban location could not be offset through restoration of a more remote or inaccessible site.

International experience

Some guidance on minimising unavoidable impacts and mitigating impacts so as to maintain the value and functionality of priority ecosystem services is given in IFC Performance Standard 6 (Box 5).

Box 5 IFC Performance Standard 6 – guidance on ecosystem services

IFC Performance Standard 6, *Biodiversity Conservation and Sustainable Management of Living Natural Resources*, January 2012, provides some guidance on the delivery of no net loss for ecosystem services. Paragraphs 24 and 25 state that:

Where a project is likely to adversely impact ecosystem services, as determined by the risks and impacts identification process, the client will conduct a systematic review to identify priority ecosystem services. Priority ecosystem services are two-fold: (i) those services on which project operations are most likely to have an impact and, therefore, which result in adverse impacts to Affected Communities; and/or (ii) those services on which the project is directly dependent for its operations (e.g., water). When Affected Communities are likely to be impacted, they should participate in the determination of priority ecosystem services in accordance with the stakeholder engagement process as defined in Performance Standard 1.

With respect to impacts on priority ecosystem services of relevance to Affected Communities and where the client has direct management control or significant influence over such ecosystem services, adverse impacts should be avoided. If these impacts are unavoidable, the client will minimize them and implement mitigation measures that aim to maintain the value and functionality of priority services. With respect to impacts on priority ecosystem services on which the project depends, clients should minimize impacts on ecosystem services and implement measures that increase resource efficiency of their operations, as described in Performance Standard 3. Additional provisions for ecosystem services are included in Performance Standards 4, 5, 7, and 8.

Habitat Equivalency Analysis (HEA) is a method originally developed for identifying the appropriate amount of compensation for interim environmental damages - such as through spills or pollution. Nonetheless, HEA has been widely applied to more long-term forms of compensation assessment, such as wetland mitigation in the USA - partly in response to weak replication of ecological functions in many offset sites. Unlike traditional economic analysis, which relates the damage costs to human use or non-use values, HEA relates to the loss of the ecological functions which underpin these values. Nonetheless, through restoration scaling, HEA can facilitate a 'function-to-function' approach for determining the amount of compensation needed to replicate functions such as nutrient cycling in an offset area (Strange *et al*. 2002). Where the relationship between local ecosystem services and ecological functions is well-understood, HEA can support replication of specific regulating and supporting ecosystem services (ICMM, 2013).

In Germany, metrics have incorporated ecosystem services relating to the environmental assets that must be assessed and if necessary offset, including air/climate (e.g. cold air flow), surface water and groundwater (e.g. groundwater recharge), landscape (e.g. aesthetic appreciation), and soil (see example in Box 6).

Box 6 Assessment of soil related ecosystem services in German offsetting

According to LUBW (2012), in Germany the soil assessment includes the loss of the productive function of soil in relation to agricultural crops or natural vegetation, the significance of the soil's role as a buffer and filter for pollution, for regulating water flows and replenishing groundwater stocks, as a habitat for species, and as a historical site e.g. for archaeology. The area affected by soil sealing is quantified and must be compensated with an equivalent or larger area that is either unsealed or restored sufficiently to compensate for the lost soil functions, e.g. through the conversion of a sufficiently large area of arable soil to permanent grassland or other vegetation.

In Baden-Württemberg, the loss of the soil's natural fertility, water cycle regulation, and pollution regulation functions are scored from 1 (minimal loss) to 5 (maximum loss) per hectare of soil lost to sealing (sealed soil is scored at 0). This gives a maximum function loss score of 4 points per ha or a minimum of 1 per ha. After subtraction of any mitigation and restoration measures the remaining score is weighed against the total score of an offset measure or measures, calculated in the same way. The score can also be translated into a monetary value using a standard rule of 1 to 5 Euros per m^2, to give a maximum monetary value of €12 500 per ha.

Signatories to the Equator Principles are usually required to implement a Social and Environmental Management System under Performance Standard 1 in order to manage environmental and social risks and impacts. For large and complex projects involving significant impacts to multiple biodiversity values, an 'ecosystems approach' is necessary and clients may be required to undertake an Ecosystem Services Review, whereby the client categorises relevant ecosystem services associated with the project and defines priority ecosystem services. Based on the review and categorisation, a client may be required to undertake further stakeholder consultation for the process of identifying priority ecosystem services.

A practical basis for implementing the EU NNL concept would be to ensure **no net loss of biodiversity at a minimum**, where possible with no-net-loss of **significant local ecosystem services in addition**. In an EU context, data and information gathered as part of the MAES (Mapping and Assessment of Ecosystems and their Services) could be particularly beneficial for prioritising ecosystem services in different geographical settings, and could significantly reduce the number of services to be mapped and assessed prior to an offset project.

Nonetheless, it is important to recognise the relative infancy of ecosystem service metrics and the difficulty of measuring many services in practice. Despite significant progress in valuation of ecosystem services, we often lack a clear understanding of what biophysical factors support services for different ecosystems and in what combinations. Without direct measurements of the processes that lead to provision of ecosystem services, or surrogate measures that have been shown to dependably represent the functions that support a service or a suite of services, there is no way of knowing if restoration activities are actually leading to the provision of ecosystem services (Palmer and Filoso, 2009).

International experience points to a risk-based approach as the most common way of incorporating ecosystem services within biodiversity offsets, whereby overall flexibility of ecosystem service trade-offs is linked to the biodiversity value of the area in question: where offsets are addressing critical habitats, ecosystem service trade-offs may be less desirable, whilst more flexibility may be possible where the species or habitat addressed by the metric is comparatively less scarce or valued.

Box 7 Mapping and Assessment of Ecosystems and their Services

Action 5 of the EU Biodiversity Strategy to 2020 calls on Member States to map and assess the state of ecosystems and their services in their national territory with the assistance of the European Commission. It recognises that achieving the headline target of halting the loss of biodiversity and the degradation of ecosystem services in the EU by 2020, and restoring them where feasible, depend on comprehensive and robust information concerning the status of biodiversity, ecosystems and ecosystem services across the EU, and the capacity to monitor changes.

A Working Group on Mapping and Assessment of Ecosystems and their Services (MAES) has been set up under the Common Implementation Framework (CIF) to support the implementation of Action 5 by the EU and its Member States. The first action of the Working Group was to support the development of a coherent analytical framework to be applied by the EU and its Member States in order to ensure consistent approaches are used. This was presented in a discussion paper in April 2013.

While MAES seeks to measure status and changes in ecosystems and their services across the EU, offset metrics are applied to specific impacts on biodiversity and ecosystem services in order to achieve no net loss at the project, organisation, sectoral, local or regional level. They therefore have a slightly different purpose, and there are additional criteria that need to be applied in analysing different approaches to offset metrics, such as their practical applicability at the project level. This may call for different types of metrics for different settings.

European Commission (2013) *Mapping and Assessment of Ecosystems and their Services*. An analytical framework for ecosystem assessments under Action 5 of the EU Biodiversity Strategy to 2020. Discussion paper – Final, April 2013

3.5.8 Ensuring equitable outcomes

Two critical issues identified above for NNL of ecosystem services are dealing with local communities and balancing changes in multiple ecosystem services in order to achieve a net NNL position and equitable outcome for all affected groups.

Incorporating stakeholders into the assessment process (as advocated by the ecosystem approach) can bring a number of advantages to the offsetting process and outputs, including:

- Using local knowledge and value systems to generate ecosystem service assessments can overcome data gaps and ensure that outputs reflect local conditions and cultures;
- The underlying causes of biodiversity loss at the offset site may be linked to unsustainable resource use practices by local stakeholders. Offering local stakeholders a viable and attractive sustainable use alternative will be key to ensure their willingness to alter existing practices in order to enable the offset outcomes to be achieved;

- Incorporating stakeholders within the process can improve local buy-in and help ensure a 'social licence to operate', with regards both the impacting operation and the offset delivery.

It was earlier recognised that it may be impossible to deliver positive changes in all ecosystem services and instead a net beneficial position should be strived for, with due regard for distributional issues. Cost-benefit analysis can be used as a framework for balancing the costs and benefits of changes in multiple ecosystem services and refining projects and offsets so as to maximise that net benefit. This can provide a perception of an overall benefit, despite the potential costs of losses from some ecosystem services. It is highlighted that biodiversity offsets are more likely to succeed where stakeholders feel that the compensation for residual impacts is providing a net improvement.

In light of this, BBOP recommends undertaking cost-benefit analysis focussed on the impacts to local stakeholders. The BBOP Cost Benefit Handbook (BBOP, 2009) provides guidance to offset planners to help ensure that:

- Local people are no worse off through the presence of the project in terms of its impact on biodiversity-related livelihoods;
- Local people at the offset site are no worse off as result of the biodiversity offsets, as appropriate and equivalent benefits are built into the offset to compensate for any negative impacts they cause; and
- Calculations of the conservation gain of the biodiversity offset activities are realistic in the assumptions made about how local people will become involved in the offsetting activities.

3.6 Conclusions on the key factors affecting the suitability of metrics and the advantages and disadvantages of different types

The descriptions of the metric types and the analysis above indicate that, with the exception of the simplest ratio metrics which are probably not fit for the purpose of a No Net Loss determination, all have their strengths and weaknesses and are suitable for use in some situations. In other words there is no single best metric or best-practice approach, and they need to be chosen according to their purpose, with reference to good practice principles that metrics, multipliers and exchange rules should endeavour to incorporate, such as ensuring they result in equity in type, space and time of biodiversity and ecosystem services. This is crucial because the success of offsets is highly dependent on the use of appropriate metrics.

Table 3.5 Summary of the main advantages and disadvantages of the main types of offset metric

Metric	Advantages	Disadvantages
Habitat (biotope) area	Very simple transparent system with low transaction costs – suitable for impacts on habitats with very low biodiversity values that do not significantly vary in condition.	Does not capture many important values of habitats. Decisions on ratios are largely arbitrary. Particular requirements for species are ignored.
Habitat (biotope) area x standard value	Relatively simple low cost system that takes into account the average potential ecological values of habitats. In combination with exchange rules allows out of kind offsets.	Habitat values can vary greatly according to their condition. Does not take size and spatial issues into account unless by a simple multiplier. Does not enable offsets that enhance habitat condition. Simple habitat metrics are not always good proxies for species requirements (particularly in low value habitats).
Habitat (biotope) area x site	Provides a much more reliable and comprehensive measure of	Does not explicitly take into account different habitat values, so can only be used

Metric	Advantages	Disadvantages
condition	biodiversity value and enables potential habitat condition improvements through restoration / enhancement to be taken into account.	for like-for-like offsets or within bands of type. Condition is difficult to define and measure, so complex methods are needed and good quality data from site surveys, which increase costs and, if poorly planned, could delay projects – so requirements are not considered reasonable for projects that clearly have low level impacts. Also less transparent and arbitrary weightings are often used for condition attributes. As above for species.
Habitat (biotope) area x standard value x site condition	Considers habitat value as well as condition so allows comparison of different habitat types and therefore unlike-for-like offsets, and offsets that improve condition of existing habitats.	Can be complex and lack transparency. Requires information on habitat values at national and local values, as well as impact and offset site data on condition. Cost likely to be similar to other metrics that assess condition. Simplified systems such as Defra metric may not be robust. As above for species.
Species focussed approaches	Often a clear, objective and transparent measure, that may link directly to conservation policies and legislation (e.g. for protected species) and stakeholder concerns (e.g. species of high cultural value).	Cannot capture many important biodiversity values without becoming highly complex – so to achieve NNL it is best used in combination with habitat metrics to identify particular requirements for important species when known to be present. Requires good spatial data and field surveys where these are not already mapped, which increases costs and can delay projects – so requirements are not considered reasonable for projects that clearly have low level impacts.
Replacement costs	Relatively simple and transparent and can make use of cost information compiled as part of a Biodiversity Offset Management Plan; particularly suitable for fee-in-lieu systems.	Costs of replacing lost habitat can vary considerably, and be difficult to assess reliably for some habitat types. Simple habitat restoration costs are not likely to be good proxies for some species requirements.
Ecosystem service specific metrics	The metrics can be chosen to ensure they are appropriate to the service and its context, thus ensuring sensitive and reliable measurements.	Data requirements are likely to be high as several services, which may be location-specific, may need to be assessed each with a different metric, and data needs for each may be significant. The use of a variety of metrics may cause confusion amongst authorities, developers and stakeholders, hindering learning, communication and interpretation of the results.
Ecosystem service valuation	Valuation (i.e. monetisation of ecosystem service changes) can enable all ecosystem services to be compared, bringing into play tools such as cost-benefit analysis which enable consideration of the	Primary valuation exercises can be financially and labour intensive and are therefore likely to be unfeasible except for the most significant of cases. The existing evidence base for value transfer is limited. Only partial valuation (i.e. of some services)

Metric	Advantages	Disadvantages
	'net' benefit of changes in multiple ecosystem services.	is therefore likely to be possible, and these estimates may not adequately reflect local variations in perceived value.

Source: further developed from NNL policy option study (Tucker et al, 2014)

Therefore, although habitat area ratio metrics take into account the value of habitats, their use should generally be avoided because they are highly reductionist and are unlikely to be able to capture biodiversity values reliably, especially if fixed ratios are set at coarse scales (e.g. nationally) because values may differ regionally and locally. Most importantly, values of most semi-natural and natural habitats vary greatly as a result of their condition or other properties such as their spatial position – as further discussed below. With such metrics there is an obvious likelihood that good examples of a habitat will be offset with basic examples of a habitat, because the latter normally have a much lower creation/re-creation cost. Thus the widespread use of such metrics is likely to result in biodiversity losses and should be avoided. However, they may be suitable for the assessment of very low value habitats such as artificial habitats, where it is more important to have low transaction costs so that workable offset schemes can be developed for them.

The use of such basic biodiversity metrics has led to one of the main criticisms of the offsetting approach being that metrics are crude measurements of biodiversity, that do not adequately capture what is important, which according to Salzman & Ruhl (2000b) is a key requirement for an offsetting metric and currency. Consequently, Walker *et al* (2009) question whether offsetting systems can reliably result in no net loss of biodiversity.

The analysis above indicates that the main approach to increasing the ability to capture key biodiversity values is to increase the sophistication of the metrics, ensuring the division of habitat types is sufficient (i.e. not too coarse) and they take into account the general potential ecological value of habitats (with respect to the habitat itself and its importance for associated species), their actual site-level condition, ecological functions and spatial issues. It is also important to bear in mind that metrics that are focused on general habitat (i.e. biotope) characteristics are not always good proxy measures for some species habitat requirements. More inclusive measures of biodiversity, and particularly important components, can therefore be obtained by including species-focussed metrics such as the HEP procedure, as for example adapted for use in Somerset, England.

It also is usually appropriate to include multipliers to take into consideration factors such as spatial issues that affect the value of impacted and offset sites (e.g. habitat patch size, ecological connectivity, the integrity of ecological processes and the viability of meta-populations of important species), risks of offset failure or low additionality, desired biodiversity outcomes for risk aversion offsets and time delays. In this respect there seems to be scope for improving the treatment of spatial issues (i.e. ensuring that offsetting properly takes into account losses and gains in terms of ecological connectivity of habitat patches and other landscape scale impacts). Risk multipliers also need to be based on empirical evidence (and as a result much greater multiplier values used than they often are currently); but not relied on as the only means of risk management.

The use of sophisticated metrics has some drawbacks, including their reduced transparency especially if numerous subjective or arbitrary judgements are required (e.g. on habitat classes and values, appropriate baselines / benchmarks for habitat condition and weighting factors). These issues can undermine confidence in the system amongst stakeholders.

The more robust metrics also need sufficient data, which often requires detailed and lengthy fieldwork by experts (especially if species are involved). The Victoria State metric and similar approaches, such as variations applied by BBOP in Madagascar, New Zealand, Sweden, Colombia and other countries (see website for case studies[14]) and HEP metrics require expertise to use, good quality data (including habitat maps if spatial attributes are to be considered) and normally field surveys of the impact and offset sites. They therefore result in higher costs than for area x value approaches in which the values are based on averages derived from existing data and no, or very limited, project-specific site surveys.

If the requirement for project-specific surveys is not planned well, this can delay projects and increase costs. Such problems, especially relating to delays, will therefore reduce the acceptance of offsets amongst businesses, especially for projects that are likely to have minimal biodiversity impacts. This problem can be exacerbated with more sophisticated systems that attempt to add in requirements for particular species and complex spatial and genetic considerations.

As a result of concerns about potential costs and transparency, some offsetting systems have developed simplified versions of the habitat area x value x condition metric, as for example in England. However, this has been criticised for being much too simple, both in terms of its treatment of habitats (Environmental Audit Committee, 2013) and its inadequate treatment of requirements for some protected species (Howard *et al*, 2013). Furthermore Somerset County Council has shown how a species-focussed metric can be nested with the simpler habitat metric to provide a more robust and comprehensive biodiversity measure (Somerset County Council, 2014). Although the approach is dependent on the availability of considerable ecological data (including standardised habitat maps, the distribution of protected species, and the ecological requirements of the species including habitat suitability indices) that are integrated on a GIS, this allows assessments of potential impacts early in the planning stage. This can help developers avoid biodiversity impacts (and associated costs and delays) and enable them to more reliably include necessary mitigation and offsetting in their business plans and project budgets. Therefore, although biodiversity survey and data management costs for such sophisticated systems may be greater than for others, given the high costs of project delays and offsetting, it seems likely that in the long-run investment in such data would be cost-effective overall and beneficial for economic development.

Lastly, the review clearly shows that whatever metrics are used they need to be carefully combined with appropriate exchange rules. This is important, because metrics do not capture all important biodiversity values and therefore a precautionary approach needs to be taken that guards exchanges in habitat type that could lead to undetected biodiversity losses. Thus exchange rules are needed to prevent high value habitats being replaced with lower value habitats (although exchanges within bands of similar value habitats may be appropriate) and areas of habitat being replaced by the same habitat type, but in lower condition (unless there is high probability that its condition will match the original habitat in a reasonable time). Exchange rules can also play an important role in ensuring important ecosystem services are maintained.

Section 5.1 draws on these conclusions to suggest possible options for metrics in the context of an EU No Net Loss initiative.

[14] http://bbop.forest-trends.org/

4 Mechanisms for Securing Long Term Conservation Benefits

4.1 Introduction

While sound metrics are required to ensure that appropriate levels of conservation activity are specified, achieving no net loss also depends on these activities being delivered in an effective, sustained and measurable way over the long term.

Experience from previous studies indicates that securing long term conservation benefits from offset schemes relies on at least three main factors:

- Ensuring the effective delivery of conservation management activities through appropriate regulatory and management systems;
- Securing the long term use of land for conservation purposes; and
- Ensuring the financial sustainability of conservation management over time.

These requirements are reflected in the BBOP Standard, which includes long term outcomes as one of its ten principles, and identifies relevant criteria and indicators to achieve this (Box 8).

Box 8 The BBOP Standard – Long Term Conservation Outcomes

The Business and Biodiversity Offsets Program (BBOP) Standard on Biodiversity Offsets sets out ten principles which together establish a framework for designing and implementing biodiversity offsets and verifying their success. Principle 8 of the Standard deals with long-term outcomes, stating that:

The design and implementation of a biodiversity offset should be based on an adaptive management approach, incorporating monitoring and evaluation, with the objective of securing outcomes that last at least as long as the development project's impacts and preferably in perpetuity.

The following criteria and indicators are specified:

Criterion 8-1 Mechanisms shall be in place to ensure that the measurable conservation outcomes from the offset will outlive the duration of the development project's impact.

Indicator 8-1-1 Evidence is provided that those responsible for implementing the offset have the requisite management and technical capacity.

Indicator 8-1-2 Legal and financial mechanisms are in place to guarantee the financial and institutional viability of the offset for at least the duration of the project's impacts, including under conditions of a sale, or transfer of project ownership or management.

Criterion 8-2 Adaptive monitoring and evaluation approaches shall be integrated into the Biodiversity Offset Management Plan to ensure regular feedback and allow management to adapt to changing conditions, and achieve conservation outcomes on the ground.

Indicator 8-2-1 Evidence is provided that the measures to manage and mitigate identified risks are implemented, the results are monitored, and that risk assessment and management are adapted as necessary throughout offset implementation.

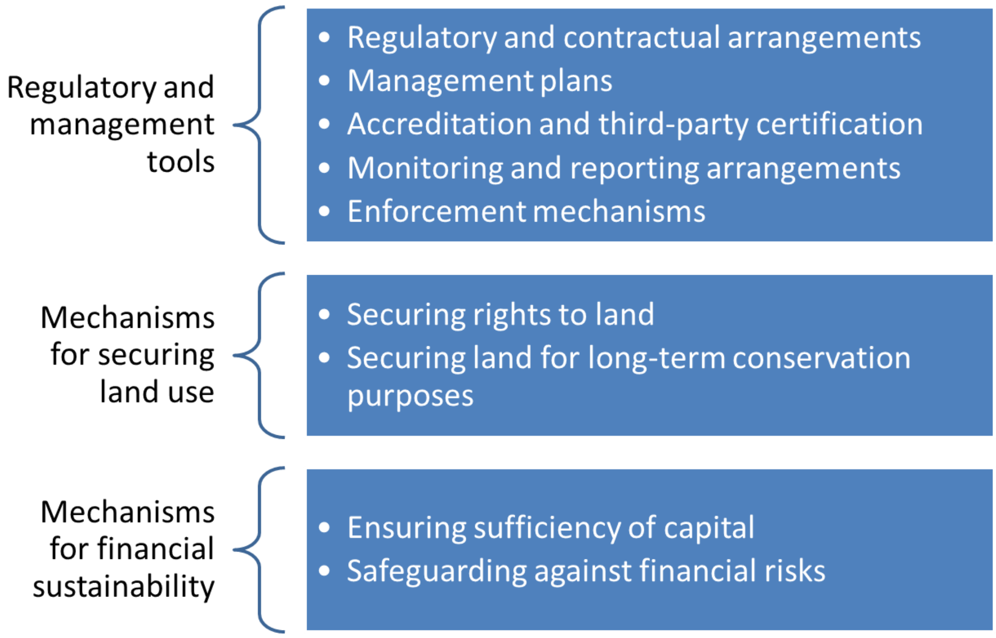
Indicator 8-2-2 Offset conservation outcomes and milestones are independently audited and project responds to audit recommendations in a timely manner.

Indicator 8-2-3 A system exists for monitoring and evaluating the success of offset implementation, including the monitoring of risks, and this provides regular feedback which is used to document, correct and learn from problems and achievements.

Source: BBOP (2012)

The three main requirements are illustrated in Figure 4.1.

Figure 4.1 Mechanisms for Securing Long Term Conservation Benefits



This section presents an overview of the individual mechanisms available and how they are currently implemented in best practice situations internationally and in the EU, and highlights whether there are likely to be any critical issues in implementing such mechanisms across the EU.

4.2 Management and regulatory systems

4.2.1 Introduction

Securing long term benefits is strongly dependent on having effective legal requirements and management arrangements in place. Before the implementation of projects, regulatory authorities need to be confident that developers will deliver the offset in a way that meets its intended objectives.

There are a several elements that can contribute to building this confidence, including:

- **Regulatory and contractual arrangements**
- **Management plans**, and associated performance criteria;
- **Accreditation and third-party certification;**
- **Monitoring and reporting arrangements**; and
- **Enforcement mechanisms**.

These different elements are guided by international standards, drawing on international experience and good practice, such as the BBOP Standard (see Box 8 above). The IFC Performance Standard 6 is also a key reference for many offset projects, as a mandatory requirement for financial institutions subject to the Equator Principles. These elements are also encapsulated in a range of national standards (such as the Australian Offset Assessment Guide) that draw together diverse provincial offset frameworks with reference to national regulation.

International experience of offsetting highlights the importance of two core elements for ensuring the maintenance of conservation benefits from offsets: robust **contractual agreements**, complemented by effective **management plans** to deliver on these contracts. Clearly, abilities to meet these requirements are unlikely to be evenly distributed across the EU, and this has implications for the ability of offsets to ensure long-term conservation benefits within a no net loss initiative.

International experience from the USA and Australia suggests that the best outcomes for long-term conservation benefits are achieved when contract design is explicitly linked to monitoring and enforcement schedules - since this approach ensures that the balance of regulatory competences at different scales are utilised and the additional administrative burden is minimised.

However, in the context of an EU No Net Loss Initiative, the responsibilities for regulating and managing offsets would be likely to differ substantially between Member States. This is reflected in the experience of delivering offsets within the EU to date, which is influenced by existing regulatory systems and institutions in place for conservation management in different EU countries.

The following sections discuss each of these elements, considering their importance for delivering long term conservation benefits, the different approaches and options to address them, and reviewing international and EU experience in their application.

4.2.2 Regulatory and contractual arrangements

Importance for securing long term conservation benefits

The delivery of long term conservation benefits depends on establishing binding agreements which ensure the continued delivery and management of the offset. Without a binding contract or regulatory requirement, there can be no guarantee that the offset will persist in the long term or will be managed in accordance with its intended objectives.

BBOP's Biodiversity Offset Implementation Handbook (2009) emphasises the need to review the legal framework or policy context prior to initiating any offset scheme. National legal and policy frameworks usually provide the high-level requirements for offset programmes to address (such as enforcement of the mitigation hierarchy, no net loss or like-for-like compensation requirements). Given the complex and specialist nature of offsets, programme-level guidance or regulation is usually necessary that outlines specific requirements and expectations for offset schemes.

Whilst initially such guidance documents were intended to provide a general guide to support compliance with national regulatory requirements, increasingly these are being formalised within contractual agreements that include explicit ecological and monitoring criteria for the site. This formalises the commitment of the regulator and developer to realise requirements such as like-for-like compensation, or in the case of a third party provider, the responsibility of this party to realise conservation benefits.

Contractual agreements provide a direct link between regulatory requirements and intended actions within the offset management plan, and ensure that implementation is binding. These agreements also provide a legal basis for enforcement in cases of non-compliance: if regulators fail to write specific criteria into the contract prior to authorisation, they have limited recourse to take action against developers who do not deliver on their commitments. They also define responsibilities on the part of the regulator (usually representing the state or a governmental body), recognising the distinct nature of biodiversity as a public good to society at large, albeit one associated with a range of private benefits.

Options for implementation

International experience of offsetting points to an array of different regulatory and legal tools to secure long-term conservation benefits, each shaped by the distinct institutional and geographical landscape in which they function. In most, but not all, cases, some element of national or provincial law provides the basis for enabling offsets and sets clear requirements. These 'offset-enabling regulations' have been seen to be pivotal in supporting demand for offsets over the long-term, as well as providing some level of assurance that offsets will remain viable under changing political and regulatory conditions (ICMM, 2013).

In many cases, offsetting is a legal requirement for development approval within established planning regulation. In this case, permits will be the key basis for the contractual agreement underpinning an offset requirement. Depending on the specific legal regime in place, the contractual agreement underpinning the offset may be a bilateral agreement between the regulator and the offset provider (in cases such as the USA, where liability transfers from developers to offset providers), between the regulator and the developer (in cases such as France, where liability is retained by the developer), and/or between the developer and the offset provider.

In recent years, guidance materials have generally shifted from providing a consultative resource to providing a more prescriptive basis for offsetting, alongside formalised contracts. This is reflective of growing awareness of weaknesses in offset implementation arising from poor enforcement of offset requirements by regulators, but is also a concession to the scientific complexity of offsetting and ecological restoration and the lack of evaluation capacities amongst local regulators, many of whom lack the skills to make judgements regarding long-term ecosystem trends or like-for-like comparisons between ecological functions.

In general, regulatory systems seek to strike a balance between requiring a sufficient level of rigour to ensure compliance with legal requirements, and allowing sufficient flexibility in terms of realising the offset in diverse regulatory or environmental settings. A common approach is to allow offsetting in the context of a wider element of national regulation, coupled with specific national or regional guidance documents describing how to ensure compliance with these regulatory requirements within a management plan. Through contractual arrangements, responsibilities and expectations can be apportioned in an equitable way between the state and the development proponent with regard to acquiring, managing and ensuring the long-term security of the biodiversity offset. This involves determining how the risks of offset failure and cumulative impacts on biodiversity should be shared between the development proponent and society at large (BBOP, 2009).

Equitable distribution of benefits between private entities, local communities and society as a whole requires agreements that balance the value of biodiversity (measured using various metrics or currencies), the value of land (measured in financial terms) and the proposed value of the development (measured in socio-economic benefits such as jobs and anticipated profit margins) (BBOP, 2009). Finding a balance between these concerns requires regulators and proponents to collaboratively design agreements that are sufficiently flexible to be tailored to specific local concerns.

Contractual agreements also need to address the legal permanence of the offset at the design stage. A legal agreement between the company, government and other relevant stakeholders needs to be developed to define the role, responsibilities and commitments of all parties. This agreement will also need to identify how responsibilities will be transferred in the event of company or organisational mergers or acquisitions - contracts need to ensure that offset requirements cannot be revoked by the new owner (BBOP, 2009).

International experience

Regulatory systems for implementing offsets can vary quite significantly between countries and regions, owing to distinct geographical and institutional influences. Nonetheless, some general lessons can be drawn regarding the interaction between different regulatory drivers.

Many countries do not address offsetting specifically in regulation, but have existing legal requirements in place relating to environmental impact assessment or urban and regional planning frameworks requiring mitigation of impacts to an extent that would be considered synonymous with offsetting (PwC, 2010). In this regard, there is a range of international experience in the delivery of these forms of mitigation, and corresponding examples of best practice in the design of regulatory and contractual arrangements so as to incentivise long-term conservation benefits. For these purposes, we regard mitigation or compensation projects that apply 'no net loss' as an operational goal to be synonymous with conservation offsets.

The **USA** has the longest experience of offset and compensation schemes in the form of Wetland Banking and Conservation Banking. These emerged in response to two major Federal and state environmental regulations (the 1972 Clean Water Act and 1973 California Endangered Species Act). Processes for assuring offset compliance with both regulations were largely ad-hoc until the issuance of Federal guidance in the 1990s and 2000s, respectively. Management of offsets is largely decentralised to the level of administrative districts for government regulatory bodies (such as the US Army Corps of Engineers or US Fish and Wildlife Service), as well as hydrological districts. Because of the specialist nature of the offsets concerned, local regulators have traditionally had significant autonomy to define the content of contractual agreements (including performance criteria) together with offset proponents or developers, subject to relevant state and federal legislation.

One outcome of this approach has been significant regional disparities in interpretation of federal regulation and correspondingly, in the requirements placed on offset providers and often weak implementation of no net loss in functional terms.

More recently, guidance for wetland mitigation has been superseded by specific regulation in the form of the (2008) Final Compensatory Mitigation Rule. These regulations are intended to address perceived issues surrounding delivery of performance criteria within existing offset projects, by providing a more defined basis for linking provision of offset credits to key performance criteria with target values identified that should mimic the trends and approximate the values of a 'reference' ecosystem.

Where reference information is difficult to reliably source, performance criteria as defined in contracts often refer to arbitrary measures informed by circumstantial reasoning (e.g. restricting invasive species cover to 2%) which may be either difficult to enforce or unrealistic in practice for dynamic ecosystems such as coastal wetlands. In contrast to standardised approaches, this has the benefit that measures may reflect local ecological conditions more closely, but it also restricts the transfer of knowledge and accumulation of know-how between sites (Quétier *et al*, 2011). Doubts persist amongst stakeholders as the capacities of local regulators to make realistic or robust scientific judgements in this regard. There is also a common perception that local knowledge, which may compensate for a lack of quantitative and measurable data, may be under-utilised.

Another key development of the Compensatory Rule is a formal preference (and credit allocation benefits) for offsets delivered through a habitat banking institution. Because habitat banks assume the legal liability for delivery of offsets, these institutions have to undergo a rigorous accreditation process, overseen by the relevant government bodies, prior to issuing credits.

Provision of 'advanced credits' (offset credits facilitating development approval prior to completion of the offset) is subject to several project design criteria and must result in the acquisition of land and physical/biological improvements within 3 years of the release of credits - multipliers are then applied to counter any residual loss in biodiversity in the ensuing period, but these are often weakly linked to ecological or biological conditions on the site - there are key concerns relating to the capacities of regulatory bodies to establish robust multipliers, that fully address residual impacts emerging from the provision of credits prior to fulfilment of performance criteria.

A number of market-based measures for biodiversity conservation and offsetting have emerged at the state level in **Australia** in response to a revision of the Federal Environmental Protection and Regulation Act (1999), which recognised the use of offsets as a tool to meet environmental protection goals. In Australia, protection of native species from a range of pressures, primarily development and urbanisation, remains a central tenet of environmental policy and this has influenced the development of offset credit systems. Nonetheless, oversetting systems between states often place very different requirements on offset providers.

Table 4.1 Major regulatory offset systems in Australia

BushBroker (Victoria)	BioBanking (NSW)
Must be used by the project developer to offset its project's impacts (if own land not suitable)	Voluntary use to offset project impacts
No online register; government acts as regulator and broker	Free online register; government acts as regulator or broker, private consultants act as brokers
Protection and management agreement for a minimum of 10 years and can be in perpetuity	Protection and management agreement in perpetuity, linked to the property title
Credit price based on supply and demand-negotiated between the project developer and owner	Minimum price based on cost of establishing and managing the site, administrative costs linked to the site and the investment returns generated by the owner

Source: CGSD (2012)

The geographical scope of offsetting is another key concern for regulatory agreements in a context of NNL. Offsetting projects in New South Wales, for example, need to demonstrate achievement of no net loss at multiple scales - including local, regional, state and national scales.

EU experience

Regulatory and contractual arrangements in the EU are heavily influenced by existing regulatory frameworks relating to compensation for biodiversity loss. Key regulatory frameworks at the European level relating to offsets are the Habitats and Birds Directives and the Environmental Liability Directive (ELD). Regulatory requirements relating to these Directives may differ substantially owing to the different focus of damage assessments; the nature Directives focus on specific damage to the Natura 2000 network and may include an assessment of baseline condition prior to development, with compensation taking place either before or during project development. By contrast, compensation under the ELD relates to impacts on specific species and can occur after the impact has occurred.

The practice of compensatory offsetting has a long history in **Germany**, and has been largely implemented at the municipal and the *Länder* (state) level, albeit with some degree of oversight by Federal government. The overarching regulatory driver is the 1976 *Nature Conservation Act* (Albrecht *et al*. 2014), with local regulation and guidelines providing more detailed clarification.

The Impact Mitigation Regulation complements the instruments specified under European law, the EIA, the SEA and the Habitats Directive Assessment. These differ with respect to their areas of application and to the legal consequences that they trigger. Whilst the SEA is carried out for plans, and the EIA at the individual project level, the Impact Mitigation Regulation is generally applicable both at the planning and project levels under the Federal Building Code and the Nature Conservation Act, respectively (see Table 4.2).

Compensation banks (also known as 'land pools') have been made possible by a 2002 revision to the 1976 law to allow additional flexibility in terms of equivalence (compensation can entail replacement of one habitat type with a different habitat within the national classification-previously full functional and habitat equivalence was required). Under the Building Code, the need for a direct spatial relationship between impact and compensation areas is partially relaxed-so compensation banks are widely applied within urban planning frameworks. This allows for a strategic use of compensation and offsets (these are used interchangeably within the Building Code) so as to maximise co-benefits for biodiversity and urban ecosystem services within long-term master plans.

Registration of compensation measures and implementation takes place at several administrative levels. The Nature Protection Department at state level publishes an update every 3 years. In many states, individual districts keep a land register and municipalities register compensation projects in a land book.

There are no official national quality standards in Germany but the German Federal Association of Compensation Agencies[15] has developed quality standards for the work of compensation agencies and the establishment of compensation pools for environmental conservation purposes[16]. Evidence from elsewhere in the EU suggests that performance standards are typically decided on a case-by-case, *ad hoc* basis. The lack of a consistent and standardised approach in many cases reflects a lack of detailed guidance as well as limited delivery experience.

Each state produces a positive list (or a negative list) of the types of projects for which offsets would be required (or is not required = negative list). This varies from state to state- for example whilst offsetting would be required for a golf course in Berlin, it would not be necessary in Brandenburg (Mau, 2012). Socio-economic and development factors thus influence the inclusion or exclusion of development activities from offset regulation. Importantly, the inclusion or exclusion of habitats, species or landscapes from the list is informed by a strategic 'pre-assessment' process for development approval of the habitat bank, which defines possible impacts on the surrounding area.

Another distinctive element of the Nature Conservation Act is its requirement for prevention or compensation of impacts on both nature and landscape conservation - thus requiring offsets to have a broad application to entire ecosystems and 'landscape scenery'. Because many of these impacts are highly subjective, this has required formal impact assessments to be undertaken even for small developments (Darbi *et al*. 2009). This broadens the remit for Environmental Impact Assessment as generally implemented under the EIA Directive- from species and habitats to impacts on biodiversity and the wider countryside- but the process of establishing descriptive criteria for offsetting is often complex and time-consuming, with significant consultation requirements (Treweek, 2009). Nonetheless, some municipalities have responded to these challenges by integrating habitat banking within existing urban development planning systems (Ecologic, 2008).

Table 4.2 Applicability of environmental impact assessment instruments at the planning and project levels (Kravchenko *et al*. 2014)

Assessment instruments	Examination level	
	Urban land-use planning (land use plan and local development plan)	Single projects
Impact mitigation regulation	Impact mitigation regulation under the Building Code	Impact mitigation regulation under the conservation law
Strategic environmental assessment	SEA Article 2, Sect 4.	-
Environmental impact assessment	-	EIA, Article 3, Sect. 1.
Habitats Directive assessment	Habitats Directive Assessment for Planning, Article 1a Sect. 4.	Habitats Directive Assessment for Projects, Article 34.

Evidence from Germany, the USA and Australia suggests that the effectiveness of contract design is likely to be heavily influenced by the scale of regulatory oversight - this in turn will be shaped by the existing institutional arrangements for conservation management in Member States, which vary widely. The key parameters that are likely to influence the design of

[15] Bundesverband der Flächenagenturen in Deutschland e.V. (BFAD)

[16] http://www.verband-flaechenagenturen.de/%C3%BCber-uns/qualit%C3%A4tsstandards/

contractual agreements in this regard are the scope of biodiversity covered by regulation (whether all elements of biodiversity, or specific priority species or habitats) as well as the level of administration of offsets (central, regional, provincial or local government).

In Germany, contractual agreements are defined firstly by the type of projects eligible for offsetting (defined at the state level within ‘positive lists’) the type of ecosystem impacted (according to standard biotope lists) the environmental context of the impact (urban or rural - with offsetting occurring under the Building Code in the former and the Impact Mitigation Regulation in the latter) as well as the scale of the impact (most small-to-medium scale impacts are addressed by municipal or regional conservation banks, whilst larger-scale impacts are often addressed by state compensation agencies or state conservation agencies).

Recent revisions to the Impact Mitigation Regulation have mandated development of clear criteria for monitoring and control responsibilities within contractual agreements, to address issues identified in previous studies of compensation outcomes (for example, Tischew *et al*, 2010, where almost half of areas surveyed had ‘poorly described restoration goals’). A lack of subsequent follow-up management was also identified as an issue for many sites.

One of the benefits of the ‘tiered’ German approach is that contractual requirements are designed to be proportionate to the initial impact on biodiversity, the degree of residual biodiversity and ecosystem service loss likely to be incurred, and, critically the capacities of responsible bodies to design appropriate contractual requirements for an offset. Offsetting in an urban environment (under the Building Code) presents very different challenges to offsetting in rural areas with higher levels of biodiversity.

In **France**, permits for offsetting and compensation represent a legally-binding agreement between the state and the developer, subject to performance criteria which may be directly linked to the offset management plan. In practice, the level of performance criteria built into the permit varies significantly between different regions of France, and these are often not elaborate or prescriptive in nature. One key distinction from more market-based systems is the assignment of liability relating to the project. Unlike US habitat banking systems, for example, where the liability for offset delivery is transferred to a habitat bank or regulatory body, in France legal liability for delivering on offsets rests with the developer and cannot be transferred. This means that the regulator can seek remediation from the developer if the project does not meet performance criteria. This creates a tendency to rely on trusted providers of offsets, or to require additional assurances or safeguards from unconventional providers.

Compulsory biodiversity offset regulation has existed in the **Netherlands** since the 1990s. The ‘National Ecological Network’ was implemented at this time to increase overall cover of nature areas from 460,000 ha to a goal of 730,000 ha by 2018. Any developments that may encroach on areas designated as an ‘NEN area' require the use of the mitigation hierarchy and biodiversity offsets as a last resort, in order to achieve No Net Loss. There are a number of laws (typically embedded within regional planning frameworks) that underpin the application of these offsets (UNEP-WCMC, 2012), but in-lieu fee arrangements (‘Groenfunds’) are the most common method of implementing offsets.

However, because actual implementation of offset measures is not rooted in a national legislation, most measures are implemented through a non-regulatory approach based on negotiation between affected parties - similar to some Canadian offset systems. A recent review of offsetting within Dutch highway planning highlighted consistent implementation of this principle within planning processes (Aragão & van Rijswick, 2014), although its ability to ensure long-term conservation benefits is perhaps doubtful in the absence of a legally-defined framework for management of compensation areas.

Offset measures have been enabled in **Sweden** since 1999 through the Swedish Environmental Compensation System. As in Germany, offsets are usually implemented by municipalities. Compensation may take a range of forms and there is a considerable level of flexibility applied to mitigation methodologies. An overriding focus on *local environmental resources* means that loss of species or ecosystems may be compensated by provision of cultural or community resources not directly linked to the impacted area.

Experience from Sweden highlights that the importance of rigour in linking high-level regulation to contractual agreements. Provisions for offsetting are included within the Environmental Code, and ecological compensation does occur with some frequency at the municipal level, but in practice there are no legally- binding requirements at the national level enforcing performance criteria such as no-net-loss. As a result, the contractual design of offset measures is left mostly to the discretion of municipal authorities, who often allow offsetting of ecological resources with unrelated resources such as cultural or educational sites.

Offsets are increasingly being applied in the **UK.** Schedule 9 ('Preservation of amenities and fisheries') of the UK Electricity Act (1989) contains the first reference of relevance to offsets in UK regulation. The Act requires generators and suppliers of electricity to preserve the natural beauty, flora and fauna, geological or physiographical features of sites of special interest, buildings or archaeological sites. Any effect that proposals would have on these features must be mitigated.

The subsequent Town and Country Planning Act (1990) strengthened the compensatory principle in UK development projects through the introduction of **Section 106 agreements**[17], which require developers to undertake specific compensatory conservation activities as a condition of planning approval in some cases. It is likely that S106 agreements will play a significant role in the financing of future offset schemes in the UK, owing to the relatively limited resources of Local Authorities to support monitoring and implementation of offsets and developer familiarity with this compensation mechanism.

Regulatory support for biodiversity offsets is also gathering pace in **Spain**, where the national government has recently ratified a new Environmental Assessment Act that can support compensatory mitigation schemes for impacts on biodiversity. Spain is considered to have significant potential for the development of offset schemes because of the high concentration of areas of high biodiversity value in private ownership (Garcia, 2012) and a range of large property developers are undertaking exploratory studies for implementing offsets under the new framework Universities and civil society groups are likely to play a strong advisory role in the development of offsets, although conservation groups are unlikely to take an active role in the management of offset sites (as in the USA) owing to a lack of financial capacities[18].

The national regulation adopted allows Spain's regions to develop their own offset requirements, which reflects the significant autonomy devolved to the regions. It is currently unclear how this will differ between regions, but the national government is shortly due to issue draft guidance addressing financing requirements, long-term conservation priorities, management planning frameworks and the use of land management agreements such as covenants. The regulatory and guidance system adopted at the national level is purposely aligned with the US framework for mitigation, with some adaptation to the specific Spanish legal context.

Possible EU implementation issues

For the design of contractual agreements governing offsets to be effective in securing long-term conservation benefits, there needs to be sufficient capacity within the responsible authority to set clear and robust objectives that will support monitoring and enforcement over the long term. In many Member States, administrative resources and fragmentation of responsibilities could present challenges to the design of effective contractual agreements. This is evidenced from experience with Natura 2000 and implementation of the Habitats Directive, where the established best practice is for regulators to define strict implementation and monitoring criteria at the permit level (Beijen *et al*. 2014).

Rega (2011) reviewed experience of ecological compensation within 25 provincial and municipal spatial plans in Italy, scrutinising the plans and associated environmental reports. Results

[17] Section 106 agreements

[18] Personal communication, David Alvarez-Garcia (02.06.2014).

indicated that a majority of the plans (66%) envisaged some form of compensation, but **most did not specify any specific binding norms for implementing** this compensation. The study then highlighted a number of barriers to the effective delivery of ecological compensation within Italian spatial planning frameworks:

- Lack of legal requirements - development approval is not linked to any obligation to deliver the compensation measures identified;
- Lack of established methodologies - there is a lack of sound, applicable methods and tools to establish the amount and kind of compensation needed or required;
- Municipal authorities pursue socio-economic considerations at the expense of ecological concerns - traditionally, the Italian planning system has prioritised these concerns, although a clear legal framework for ecological compensation could prompt the necessary change in attitude amongst policy-makers.

In these cases, revisions to the existing regulatory framework (as in Germany) may be necessary to establish broad performance principles for any offsetting system and assigning clear responsibilities for various authorities to coordinate in the design and implementation of contractual agreements.

Effective contractual arrangements need to be underpinned by:

- Robust data collection activities;
- Clear assignment of responsibilities;
- Clear regulatory expectations; and
- Appropriate institutional arrangements.

Contracts can also be used to enforce performance standards, in particular to strengthen the ecological rigour of implementation, maintenance and monitoring processes. In Germany, the 'measures sheets' provide a basis for ongoing management requirements and planning approval.

Voluntary standards - such as the BBOP Standard - can be a useful resource to establish minimum performance standards at the national or regional level. This would streamline the regulatory assessment process and integration of datasets.

4.2.3 Management plans

Importance for securing long term conservation benefits

Robust management plans are essential to ensure the provision of conservation benefits over the long term, providing the basis for delivering on contractual or regulatory agreements. For the purpose of securing long-term benefits, it is essential that those engaged in the development of the management plan have the necessary capacities and expertise (ecological, financial, access to local knowledge) to deliver on intended outcomes over the long term. No planning frameworks are universally applicable and it is important that plans are drawn up through close cooperation between the regulator and offset provider.

The design of offset projects is technical and comparatively short (often weeks or months) whereas the implementation of offsets is practical and can last several decades or longer. Effective management plans should take a long-term perspective, building in flexibility for changing external conditions (financial, political or environmental). In many cases, access to this information may be beyond the capacities of regulatory authorities and offset proponents may need to seek specialist expertise through accredited specialist consultancies or habitat banking institutions with access to specific expertise in addressing such issues.

Options for implementation

BBOP (2009), in its Biodiversity Offset Implementation Handbook, emphasises that developing an offset management plan is a key step in identifying required management resources and assigning responsibilities, and therefore in helping the offset to achieve its objectives. It states that the plan should identify:

- The management objectives of the offset;
- The activities and outputs necessary to achieve these objectives, and their timing;
- The resources (funding, technical expertise, etc.) to carry out necessary activities and produce outputs;
- The roles and responsibilities of the actors and stakeholders involved;
- The assumptions and risks in implementation, and how they will be addressed; and
- Arrangements for monitoring and adaptation to changing conditions.

Regulatory bodies are increasingly demonstrating a preference for specific types of offset delivery - an example is the US Final Rule on wetland mitigation, where accredited mitigation banks are the preferred method of delivering an offset due to their specialist expertise and capacities in planning and implementing offsets.

BBOP (2009) suggested management of the offset will proceed more quickly and smoothly when existing institutions with conservation experience can be identified to play leading roles. The challenge here relates to identifying appropriate institutions and clearly specifying their roles and responsibilities in determining which institutions may have the necessary capability to provide the desired level of management. In addition, the chosen structure may require some level of coordination amongst various institutions. The management plan is the natural stage to engage these institutions, since setting robust and valid objectives and goals is essential to ensuring effective performance criteria.

Adaptive management approaches are important in enabling the goals or intended outcomes of the offset to be realised under such changing external conditions (RedLAC, 2011). Experience from strategic conservation planning points to the benefits of engaging local stakeholders at an early stage in the production of a relevant management plan, particularly as these groups may have access to information relating to long-term species trends. Even where direct consultation with stakeholders is not possible, some degree of analysis of local drivers of change is important. Management actions may be threatened, for example, if adjacent landowners decide to subsequently alter their land management practices.

International experience

In the USA and Australia, it is common practice to require management plans as a means of setting the requirements for offset contracts. Nonetheless, there are often practical issues in the design of these plans.

In the USA, mitigation banks are regarded as the first preference by US CoE regulators under the Final Rule because of their specialist capacities and the comparative ease of evaluating offset credits. Discussions with US-based stakeholders suggest that *financial planning capacities* are often a key gap within current management planning because most professionals working in both mitigation banks and external consultants have a natural science background and only limited understanding of the complex financial assurances and incentive systems needed to ensure the viability of offsets over the long term- respondents cite typical cost analyses for wetland mitigation projects over 50 pages in length. Furthermore, project assessment reports often apply limited scrutiny to surrounding environmental or economic factors and local knowledge.

Australian offsetting systems emphasise the benefits of drawing on multiple levels of information when developing offset management plans. The national Australian Offset Assessments Guide, for example is intended to complement existing state-level planning systems so as to harmonise compliance with the Environmental Protection and Biodiversity Conservation Act. The accompanying guidance material highlights project-level biodiversity management plans as one data source in a chain of planning materials, including state-level offset management plans and biodiversity management strategies, emphasising the need for ensuring coherence between the goals of these different management plans. For example, state-level biodiversity management *actions* provide high-level regulatory requirements that can inform management plans at the regional or project level, whilst state biodiversity strategies provide access to data relating to long-term ecological trends and threatened species that can inform parameters for management planning.

EU experience

In the EU, providers engaged in the development of offset management plans benefit from a mature institutional and data infrastructure relating to conservation management planning, particularly with regard to Natura 2000 areas. Integration of knowledge from these activities within offset management plans can strengthen the reliability of performance criteria and ensure integration alongside strategic conservation goals.

In Germany, for example, responsibilities for offset management plans are primarily undertaken by the municipalities managing the habitat bank, who have access to detailed knowledge on local environmental trends as well as economic development patterns. Parameters for offset implementation are then developed in accordance with these data. **One of the particular strengths of the German impact compensation is its integration within a range of parallel policy areas**. Management planning has to relate to other activities, such as traffic, housing, nature, species and recreational activities. Integration of the compensation regulation within other federal regulation, such as the German Building Act (1998) has been made possible by the development of compensation measures within urban and regional planning frameworks at the municipal and state level. Land pools have also helped to ensure the coherence of projects from local to regional level (Rundcrantz & Skarback, 2003).

In the municipality of Mainz, for example, the local authority has implemented its own habitat bank within the urban development planning system. In this context every building area has to be associated with a remediation area - these are then pooled together to ensure coherent planning of green space within the municipality. This results in a range of social/amenity and environmental benefits (e.g. temperature or flood risk reductions) whilst contributing to no net loss goals, and can be seen as a creative application of ecosystem service approaches to offsetting within an established regulatory system.

Box 9 The BioCom project- Dutch Companies' experience with biodiversity compensation

Representatives from the business community, government and non-governmental organisations participated in the Dutch BioCom initiative, which was financed by the Ministry of Environment. The focus was how biodiversity compensation could be developed from a business perspective with the aim of developing a practical approach to project development and management planning. This built on the approach of BBOP whilst also considering the impact of supply chains on biodiversity, and the need for compensation of these impacts.

The Dutch Government, two NGOs (HIVOS, Wetlands International) and three companies- BioX Group BV (energy), Kruidenier Groep BV (foodservices) and Koninklijke Houthandel G. Wijma Zonen BV (timber) worked together to develop biodiversity compensation plans for the companies involved. This resulted in four options for organizing and managing future compensation initiatives: using existing systems or initiatives; outsourcing execution and management of the drafted compensation plans to a third party; the company executing the compensation plan; and finally setting up a new compensation system (e.g. a habitat bank).

BioCom raised a number of interesting discussions regarding supply chain responsibilities and boundary setting, indirect effects and historical loss, which have a strong relevance in the context of the No Net Loss initiative. Summaries of the three compensation plans highlight the idiosyncratic nature of compensation requirements for companies, as well as the complexity of addressing biodiversity impacts in the supply chain. The rules of compensation cannot be clearly defined and management plans cannot be clearly defined before engagement with NGOs and other stakeholders.

One general conclusion was that investing more efforts into supply chain compensation (in particular, development of tools, undertaking of research or execution of additional pilots) will help drive uptake of private sector biodiversity compensation, making it a valuable tool to motivate or press companies into conserving and sustainably using biodiversity and taking a proactive approach to managing supply chain risks (de Bie *et al*. 2011)

The effectiveness of private sector-led measures is evidenced by the approach taken to offsetting within regional planning structures in the Netherlands. This points to benefits for job creation, infrastructure investment and benefits for habitat.

Experience from Germany (and elsewhere in the EU) points to the crucial role of consultation and stakeholder engagement 'on the ground', as well as some of the benefits of greater flexibility for offsetting in urban areas. Management plans are only likely to be effective in delivering long-term conservation benefits if they are sympathetic to the local social, economic and environmental context, have support and 'buy-in' from key groups such as landowners, and meet certain feasibility criteria over the long term- including realistic management and capital investment costs.

Nonetheless, excessive influence of specific stakeholders in the design and implementation of offset management plans may in fact jeopardise the provision of long-term conservation benefits from these areas (by placing undue emphasis on economic returns from management activities at the expense of conservation needs) and may add significantly to the time and costs of delivering offsets.

One of the particular strengths of management planning of offsets in Germany is the consideration of the wider landscape context in the development of management plans. This is particularly the case with regard to accreditation processes for conservation banks, which need to develop a broad strategic plan that is aligned with local regional and conservation planning frameworks. In line with best practice in New South Wales, this requires that offset areas are planned strategically so as to maximise opportunities for ecological coherence and to anticipate future trends that may impact on the viability of the offset (for example, development pressure or climate change).

Elsewhere in the EU, management planning for offset sites typically occurs on a case-by-case basis, often drawing on input from specialist ecological consultancies, and there are few examples of strategic management planning in line with German or Australian experience. This could be seen as a gap in existing approaches, because external stakeholders are likely to influence the management of the offset over time, if indirectly.

Possible EU implementation issues

Widespread delivery of biodiversity offsetting across the EU as part of a No Net Loss Initiative would introduce a requirement for offset management plans, and be dependent on sufficient capacity to deliver them. Capacities for management planning relate to the organisation and skills of civil society, public and private sectors in respective Member States and can thus be distinguished from the ability to deliver conservation benefits through regulatory systems alone. More so than regulation, there would appear to be a significant divergence of management planning capacities across Europe.

Evidence from Member States points to a generally mixed performance with regard to management planning of conservation sites. For example, while management plans are required for all Natura 2000 sites, progress in developing such plans has varied widely between Member States (eg. Snethlage *et al*, 2012; Gantioler *et al*, 2010). To the extent that this reflects variations in capacity across the EU, it may present challenges for the widespread development of management plans for biodiversity offsets.

4.2.4 Accreditation and third-party certification

Importance for securing long term conservation benefits

Accreditation of offset providers (i.e. conservation banks, land pools or specialist consultants engaged in the design and implementation of offsets) and/or certification of offset sites as meeting requirements (by regulators or recognised third-party certification bodies) can help to verify the ability of offsets and offset providers to realise term-conservation benefits.

Using an **accredited supplier of offsets** can help to build confidence in offset provision, provided the requirements are suitably rigorous. This can be important where offsets are being conducted voluntarily, and particularly in cases where liability for the offset remains with the developer even when the offset is being delivered by a third party.

Similarly, future offset proposals may be affected if there is little evidence to show that an existing offset is achieving its stated objectives. A trustworthy, independent arbiter can be important to verify the outcomes of a specific project, or to certify that a developer/provider is able to, or has in the past evidenced that it can, deliver on its promises. **Certification systems** help to build confidence in offset provision, particularly for providers intending to engage in a multitude of transactions, for example through habitat banking. Engaging in a transaction with a certified supplier enhances confidence that offset requirements are being adequately met. There are also benefits to the developer and/or provider, in terms of its license to operate and/or reputational advantages (BBOP, 2009). Developers are increasingly influenced by the investment community who view certification systems as a key element of good practice in the management of environmental and other risks (Walker and Howard, 2002).

These measures are relevant in that some of the requirements can relate to securing long term benefits. For instance, in Germany, the German Federal Association of Compensation Agencies has developed quality standards for the work of accredited compensation agencies and the establishment of compensation pools (i.e. habitat banks) for environmental conservation purposes, which include, *inter alia*:

- Safeguarding areas and measures over the long term;
- Monitoring and follow up of the development of the pool areas; and,
- Compliance with high performance standards.

Options for implementation

Assessment of the historical performance of offset proponents and providers is assuming increasing importance in many market-based systems such as Wetland Banking as a risk-benefit based management strategy, and is a key assessment criterion for wetland credit release schedules under the revised Compensatory Rules. Developers with a strong track record of offset implementation, or who take steps to verify their existing efforts transparently, may be able to benefit from advanced provision of credits as well as more favourable collateral or insurance requirements.

One element of accreditation that is drawing increasing interest in Australia is the use of third party assessors (consultancies, academic institutions) in the design, rather than monitoring phase, of offset schemes. This draws on specific lessons learned through agri-environmental conservation schemes in Australia, and highlights the importance of incorporating specialist expertise at the earliest stage of the project, so as to maximise potential ecological and financial efficiency of the project. Experiences from Long Term Ecological Research (LTER) programmes emphasise the importance of scientific expertise in the design of monitoring programmes (the selection of parameters and development of the sampling design- where, when and how to sample as well as details of the statistical design), as well as periodical assessments of the monitoring programme in the form of adaptive management (Franklin *et al*. 1998).

Box 10 Third party design of conservation schemes in Australia

In Australia, a recent federal government inquiry relating to the harmonisation of state offset design and establishment of best practice within the Offset Assessment Guide[19] formally recommended that third parties should be involved in the design of offset projects, specifically third parties with capacities and scientific expertise necessary to design effective offset projects tailored to the specific environmental context of each project. This approach is informed through strong experiences of third-party design of conservation projects through competitive tendering systems in states such as Victoria (specifically the BushTender agri-environmental schemes). Following this established model, parties such as specialist consultancies as well as academic institutions could bid for proposals of offset sites, which could then be entered into habitat banking systems.

The rationale for this approach lies in the rigorous scientific nature of offset systems, coupled with the frequent lack of capacities amongst government bodies and regulators. The highly idiosyncratic nature of project-based monitoring projects makes involvement of scientific experts particularly critical at the project design phase (Franklin *et al*. 1998). Specialist designers of offsets can draw upon additional expertise in restoration ecology and statistical modelling processes to 'build-in' a range of risk profiles to the project. One best practice example was a winning proposal by an academic team at the Australian National University for a project under the Australian Government's Environmental Stewardship Programme. The proposal utilised a Bayesian model (a statistical probability model, usually informed by a range of expert judgements) and predictive outputs are thus able to be refined using the most reliable and up-to-date information available relating to ecological dynamics, socio-economic factors and other variables.

Certification mechanisms for offsets may be integrated into existing systems, for example upgraded ISO14001 or Mining Certification Evaluation Projects (Burgin, 2008). Studies exploring the implementation of Conservation Offsets in Alberta, Canada (Dunn and Raven, 2012), emphasise that established criteria for the development of Carbon Offsets could be applied to Conservation Offsets and considered equivalent to specialist environmental management systems for carbon such as ISO14064.

International experience

In the USA, the process for accreditation of a new mitigation bank is strictly regulated, and is usually specified through a bilateral agreement between the bank and state authorities. The accreditation process appraises the ecological performance of the bank, subject to a performance obligation, and its financial viability. The elements assessed are the location and mechanisms for locating the site, the ecological actions, the anticipated performances, their monitoring, the rules for assessing ecological losses and gains and financial guarantees. Once accredited, the bank can begin selling offset credits. The rigour of this process is essential as it underpins the transfer of legal liability from the developer to the bank.

EU experience and issues for wider implementation

Accreditation is used in some settings and is beginning to be explored more broadly as an option in the EU. In Germany, for instance, compensation pools and agencies are accredited if they fulfil a series of nature conservation criteria. The use of an accredited pool or conservation bank can reduce the amount of compensation required because of the lower risk profile associated with the project.

[19] The Offset Assessment Guide is a key element of the Environmental Protection and Biodiversity Conservation Act (2007), utilising a balance sheet approach to quantify impacts and offsets. It applies where the impacted protected matter is a threatened species or ecological community, and is a tool that has been developed for expert users in the department to assess the suitability of offset proposals.

Use of accredited offset providers is not evident in other EU countries, although there are some indicative steps towards accreditation - for example, in France. Accreditation is likely to assume increasing importance as habitat banking develops in France, particularly as liability for offset delivery remains with the developer under French law, even when the offset itself is provided by a third party. At present, accreditation of habitat banks in France is only possible if the bank possesses a trust fund whose annual interest payments allow it to finance the management measures throughout the entire commitment period (CGSD, 2012). This illustrates the role that accreditation can play as a means of delivering other measures to achieve long-term sustainability of offsets - in this case addressing financial security.

Accreditation and certification could play an important role in the development of offsetting systems elsewhere in the EU, helping to ensure that standards are in place to secure long term conservation benefits. However, limited experience of offsetting in many Member States would provide challenges for certification and accreditation.

4.2.5 Monitoring and reporting arrangements

Importance for securing long term conservation benefits

Established **monitoring and reporting** procedures are essential for ensuring long-term compliance and transparency, given that successful offsets require the sympathetic management of habitats over time. Moreover information gleaned from continuous monitoring can highlight aspects that need to be adapted and improved as the offset is being delivered if needs or circumstances change (i.e. adaptive management), which can ensure that the benefits that are delivered over the long term are maximised. Offsets that are independently monitored and/ or audited are usually regarded as more trust-worthy than those that are monitored and verified by the developer or provider itself.

Options for implementation

A long-term (management in perpetuity) monitoring perspective requires a number of safeguards to be built into the design of offset monitoring systems to ensure the viability of the scheme in the face of changing external conditions. Experience from regulatory offset systems demonstrates that the major issue with most regulatory offset systems is compliance, which is related to weak or insufficient monitoring capacities, even where the initial offset management plan appears to be suitably robust (ICMM, 2013; Burgin, 2009).

A practical concern relates to *what is being monitored* - in many cases, monitoring may relate to key species or habitats (owing to high-level regulatory drivers such as national Biodiversity Action Plans) but pay insufficient regard to the maintenance of associated ecological functions and supporting ecosystem services between the impact site and the offset area. Evidence from offsets targeted at priority or threatened species in France suggests that this can lead to an overall reduction of species richness in the order of 5-10 times that of the original impact site (Regnery *et al*. 2013). Effective monitoring systems should consider a broad range of species within the impact and offset sites, as well as the ecological functionality of the site as an integrated landscape unit.

Adaptive management and capacity-building should be elements of any long-term monitoring system. Whilst the development of the offset site may last for a period of weeks or months, some formal system needs to be in place to ensure the viability of the project over longer timescales.

Management planning issues are complicated by the fact that monitoring is unlikely to be hierarchal - neatly structured monitoring programmes in which various parameters are nested within a common sampling design are unlikely in practice. Rather, parameters may overlap or occur at different points in space or time (Lindenmayer *et al*. 2012). Approaches such as double-sampling - analysing the same location using different methodologies - can give valuable insights into internal ecological processes.

Above all, it is important to match clearly-defined objectives and performance criteria with monitoring of outcomes through an *explicit approach for determining the scope, scale and nature* *of conservation activities needed* (BBOP, 2009), especially as different parties may evaluate project success differently (Bull *et al*., 2013). The Basslink marine pipeline project in Australia provides an indicative example - this project was managed for net gain in native vegetation, and has been lauded as a successful international example of offset success by organisations such as BBOP. Nonetheless, other studies of the project have concluded that the overall impact of the project was negative, with offsets not achieving perceived project objectives (Duncan & Hay, 2007).

International experience

Access to high-quality and preferably long-term data relating to species and habitats is essential to ensure the validity of no-net loss claims. Biodiversity monitoring data in the form of species richness and species abundance datasets can provide valuable information at the level of landscape planning, whilst relevant Biodiversity Action Plans provide an important tool for 'framing' monitoring questions and making efficient use of data.

Having such data infrastructure in place can be valuable for defining a baseline for no net loss that reflects predicted landscape trends. In Australia, for example, native grassland is deteriorating in response to invasive species, so managing grassland to prevent further degradation could deliver a net gain against a baseline that incorporated landscape trends (Gordon *et al*. 2011). This is a different form of additionality to active habitat creation, which occurs against a fixed baseline. Bull *et al*. (2013) argue that no net loss should be defined against a baseline that incorporates dynamic trends in species and landscapes, since this is ostensibly the case for European environmental impact assessment legislation

Many offset schemes implemented to date have attempted to reach no net loss through defined ecological baselines that do not account for ecological trends or social values associated with the species or habitat that is being impacted. In US Wetland Banking (and to a lesser extent, Conservation Banking) an overriding focus has been on the maintenance of priority species against a baseline that does not account for ecological trends or service values associated with the local wetland area. As a result of this, monitoring has often focused on these species as a proxy indicator for other ecological functions, services or species (see Table 4.3). As a result of this, no net loss has typically been realised from a species conservation perspective whilst valuable ecological functions have been weakly replicated, if at all, within offset areas.

Table 4.3 Types of monitoring methods for US Wetland mitigation

Type of method	Simple evaluation	Partial tailor-made evaluation	Exhaustive tailor-made evaluation
Description	Measures a characteristic that is quick and easy to observe and acts as an indicator for one or more ecological functions or services	Directly measures a function	Quantitatively measures a set of functions based on numerous observable characteristics
Examples	Surface area, number of species	Percentage of duck habitat, standard of water purification	
Percentage of use	**53%**	42%	5%

Source: Duke Law School (2005)

One less-discussed aspect of US mitigation banking is that authorities (particularly US Corps of Engineers officials) tend to focus their monitoring efforts on the activities of mitigation banks rather than individual offset projects. Nonetheless, resources for this oversight remain inadequate - a study conducted by the National Research Council identified that 63% of banks were inadequately monitored (Eftec, 2010). In an attempt to remedy this problem, in 2010 the national authorities created an online database (Regulatory in-lieu fee and bank information tracking system, RIBITS). This lists the location, surface area, status, governance and type of credits associated with each bank. The database also contains information about existing credit categories and methods commonly used for evaluating losses and gains for each state and is used to monitor the geographical cover and scope of land use for offsetting.

Assessing the historical performance of habitat banks or offset programmes as a whole through such centrally-held registers can provide a useful tool for monitoring and would aid understanding of the long-term effectiveness of offset schemes (Bull *et al*. 2013).

Stakeholders in Australia also concur that programme-level monitoring can be a more efficient and effective means to assess implementation of offset programmes and compliance with regulatory requirements than in-depth evaluation of individual projects. The BioBanking programme, for example, has an accessible online database of offset areas entered into the scheme, which can be used for monitoring purposes as well as linking buyers and sellers of credits. One requirement of monitoring under the BioBanking scheme for offsets has been that no net loss must be demonstrated at multiple scales (local, regional, state and national), i.e. relating to both project-level to programme-level impacts: having a centralised database of offset areas eases the comparison of impacts across these various scales.

Box 11 Ecological monitoring of Australia's Environmental Stewardship Programme

Some strong examples of long-term monitoring considerations have emerged within Australia's Environmental Stewardship Programme. Development of 'fit-for-purpose' social and ecological monitoring systems is a core element of the Environmental Protection and Biodiversity Conservation Act (1999) and the Australian Government has engaged with the scientific community in an attempt to strengthen monitoring processes for conservation programmes and offset programmes. Processes developed for monitoring the Environmental Stewardship Programme have also been widely applied to the monitoring of offsets in a number of states.

One example of this engagement with the scientific community is outlined by Lindenmayer *et al* (2012) in their analysis of the Environmental Stewardship Programme and its associated monitoring processes through a case study of grassland conservation in south eastern Australia. Amongst other findings, the study emphasised the importance of designing tailored monitoring systems to match specific project objectives and programme size- a key element of this was adapting the monitoring regime to the specific experience and capacities of the surrounding agricultural communities (who helped inform the baseline assessment and understanding of local ecological trends).

The evaluation established a strong link between the overall cost-effectiveness of monitoring programmes, and the scientific integrity of the programme over time. The study highlighted flexible ways to reduce monitoring costs whilst still maintaining monitoring capacities, by building these considerations into project design from the outset and drawing on an understanding of the local context.

One key gap relating to monitoring processes is the timing of monitoring and reporting, and a weak alignment with ecological timescales. Evidence from ecological restoration suggests that whilst some ecological indicators, such as biomass and species richness, may restore rapidly, other important ecological functions such as species composition and food webs, nutrient cycling and soil chemical processes, may take decades to recover. Since offset schemes are typically monitored for a defined period of time, this may create a misleading representation of net loss/gain because these long-term functions fall out of the temporal scope of observation (Maron *et al*. 2012)

In US Wetland Mitigation Banking, US Conservation Banking and NSW Biobanking, offset providers are required to monitor their performance against standards established in the offset or banking agreements, and to submit monitoring reports periodically (OECD, 2013). Defined monitoring cycles are of particular importance in the context of such market-based systems because of the inherent incentives for providers of offsets to ‘cut corners’ in amounts of mitigation investment (thereby minimising overall costs), as oversight by the regulatory authority is not continuous.

Regulators have a number of means to minimise these risks, including more regular monitoring, and detailed ecological assessments- but some solutions run the risk of increasing the cost of contract enforcement so as to negate any social surplus from the offset scheme (Hallwood, 2007). Considering some of these concerns explicitly from the outset of programme design can allow regulators to respond to these challenges in a more cost effective manner.

EU experience

Although in principle there are monitoring and reporting requirements for offsets being implemented in the EU, research by ICF GHK (2013) on habitat banking and IEEP *et al* (2014) on policy options for delivering no net loss suggests that in practice this is an area that is often lacking and is in need of considerable improvement.

The performance of regulators in scrutinising the maintenance of long-term benefits from offsets is again likely to be somewhat uneven across the EU. A key issue may be the fragmentation of regulatory authority for different environmental media in some countries.

In Germany, it is widely recognised that there has been insufficient monitoring of offsets in the past, mostly due to a lack of clear obligations and complex and confusing requirements under the Building Code and the nature conservation legislation. This weakness has been recognised and partially addressed by strengthening of the Impact Mitigation Regulation - under Article 17.7 competent authorities are now required to review whether the required project mitigation measures and offsets-including maintenance measures- are carried out properly and on time. The authority may also require the intervening party to provide a report.

Planning officers within competent authorities have a key role in ensuring the offsetting objectives are clear from the outset (within the design of contractual agreements) so that it can be reliably established whether an offset has achieved its objectives. They also need to check outcomes, for example by obtaining a certificate of completion from the offset provider (Tucker *et al*, 2014).

Many German states have developed sophisticated systems for long-term monitoring and evaluation within so-called *measures sheets* (Annex 1.4). These measures blend prescriptive criteria for reporting of project results with specific monitoring requirements for legally-protected biotopes created as part of the compensation process. Suggested monitoring goals are adapted to specific habitat types and provide guidance on the nature of the monitoring to be undertaken. Some examples of monitoring measures for impact assessment and project design are included in Annex 1.4.

The guidance also includes recommendations for implementation control (monitoring management aspects of offset creation) and functional control (monitoring changes in ecological condition arising several years after project completion). A number of the project reporting criteria relate to the linkages between management control and functional control, such as the requirement to juxtapose landscape care measures with possible impact scenarios. As such, the monitoring guidelines provide a template for the development of adaptive management systems by blending ecological criteria with management flexibility ‘on-the ground’ (Kravchenko *et al*, 2014).

Sachsen-Anhalt is the only federal state to have passed a regulation that legally obliges impact monitoring of offsets (Bruns, 2007). Offset monitoring is the responsibility of local authorities in the urban zone and regional nature conservation authorities in the rural zone. However, local authorities often lack the capacity to carry out monitoring of the biodiversity and conservation objectives (Steffen 2007).

In France, responsibility for inspection and monitoring is contained within the Environmental Code, reporting of monitoring and enforcement measures is fragmented according to different media or localities, which increases the regulatory burden of monitoring and limits exchange of best practice. However, a planned 'offset gateway' will provide a single resource for these reports in the future (GCSD, 2012).

Possible EU implementation issues

One issue in monitoring the performance of offsets in an EU context is likely to be the aforementioned fragmentation of regulatory authority and a lack of clarity in who is responsible for verifying that contractual agreements are fulfilled, and when. In contrast to the evolving system of monitoring and verification in the USA and Australia, financial penalties for non-compliance with contractual agreements appear to be rare - this is linked to the infrequency of monitoring and enforcement.

In Member States with a comprehensive Environmental Code, requirements for ongoing monitoring and verification of offsets are likely to be required in law. In others, such as Germany, the lack of ongoing monitoring is seen as a deficiency within existing offset frameworks. Monitoring should normally be the responsibility of the developer/offset provider, according to the polluter-pays-principle, but there is also a need to ensure that these responsibilities are routinely met.

In any case, stepping up regulatory oversight of monitoring and verification activities could entail significant administrative costs and there is a risk that the regulatory burden of monitoring offsets may result in the reallocation of environmental expenditure from other areas. Ideally these costs would be recouped from offset proponents through an administrative fee (as in NSW BioBanking), although this would clearly add to the overall costs of offsetting.

One approach would be to only permit offsetting by accredited providers, so the responsibility for monitoring would still lie with the developer/provider. This could be specified within the accreditation/certification process and the provider's management plan.

In the USA, this is the 'favoured' approach for mitigation banking because banks go through a rigorous accreditation process (as do German conservation banks). However, this does not replace the need for some public oversight. Conservation banks in Germany are sometimes disadvantaged by higher costs in comparison to direct offsetting by developers - but arguably this situation has arisen because the implicit costs of administration and monitoring (where these occur) are absorbed by public authorities.

In each case the regulatory tools outlined in earlier sections- such as contracts and management plans - provide a means of specifying and formalising monitoring requirements.

4.2.6 Enforcement mechanisms

Importance for securing long term conservation benefits

Enforcement mechanisms are actions which can be taken against the developer or permit holder in the case of a breach of contract or a breach to the agreed conditions on delivering the offset. They are required to ensure that actions are appropriately and effectively carried out, particularly where they are a condition of planning approval, permits or project finance. The ability for relevant bodies to discharge their enforcement obligations is linked to the efficacy of legislation and the financial and resourcing capacity of regulating bodies. Enforcement actions are often based on the gravity of the fault.

Options for implementation

Enforcement mechanisms will inevitably be informed by the specific regulatory regime in place for the offset and the geographical scope of offset administration. Enforcement actions can include:

- Written warnings;
- Infringement notices or fines; and,

- Court actions such as prosecution.

Enforcement mechanisms should go hand-in-hand with effective monitoring systems. The first point of reference for any enforcement system should be a defined **contingency plan**. This is a requirement for regulatory approval under US Wetland Mitigation, for instance, and identifies the measures that will be taken should projects fail to meet defined milestones or performance criteria. Where credit-based offsetting systems are in place, restricting the supply of credits is the next natural step. Where these steps are ineffective or non-applicable, more formal enforcement mechanisms would be deemed appropriate.

International experience

In Australia, all tiers of government allocate resources for compliance and enforcement activities. Moreover, financial penalties and criminal convictions can be imposed on breaches of environmental legislation, which includes the legislative framework for offsets and habitat banking. Development which proceeds without approval can attract both criminal penalties (up to 7 years imprisonment) and financial penalties up to €4.6 million.

The US also imposes administrative, civil and criminal penalties with administrative penalties that can reach almost €130,000 and civil penalties imposed in a judicial proceeding can reach €26,500 per violation per day (ICF GHK, 2013). Nonetheless independent assessments point to many cases where actual enforcement has been limited by the tendency of assessors from governmental agencies to resolve noncompliance through renegotiation of performance milestones or criteria rather than penalties (US GAO, 2005). This may be symptomatic of the decentralised nature of offset monitoring and the lack of clear mechanisms for enforcement of standards at the federal level.

EU experience and issues for wider implementation

Experience from the EU indicates that mechanisms to enforce conditions are not always included and there are rarely penalties for non-compliance. In some countries (e.g. Sweden) this element of the system works relatively well in that developers are held accountable for the outcomes of the compensatory measures. In countries such as Germany and the UK, examples of enforcement appear to be more limited. In Australia and the US enforcement of conditions is more prevalent, although the use of legal exemptions is reported to be widespread and, where conditions are not built into original agreements, regulators often have little recourse to take action against projects that do not meet requirements of ‘like-for-like’ restoration.

In Germany, the planning and decision-making authority is legally obliged to ensure that offset measures are carried out and can demand a report from the developer[20]. However, it can only legally enforce the measures specified in the planning agreement (Breuer, 2010). It cannot legally enforce control of whether the offset has actually achieved the restoration of biodiversity and ecosystem functions it was supposed to achieve, unless such a clause has been written into the planning agreement.

Whilst it is generally recognised that monitoring and verification of compensation measures undertaken in accordance with relevant legislation in Germany is essential, in practice the scope of enforcement activities relates closely to the scale and impact of the development in question. Many municipal habitat banks and planning authorities deal with thousands of small-scale mitigation measures annually, and detailed monitoring and verification of these activities may be neither feasible nor appropriate. Verification checks by competent authorities are mostly limited to random checks or regular checks against large or risky offsets (in which case, the state conservation authority is usually responsible). Sachsen-Anhalt is the only state which legally obliges impact monitoring of offsets (GCSD, 2012).

As with other areas of environmental policy, wider implementation of biodiversity offsetting in the EU, in the context of the No Net Loss Initiative, would clearly be dependent on the development

[20] Defined in §17.7 BNatSchG 2009

of effective regulatory and enforcement systems. This would depend on sufficient capacity and expertise among the relevant regulatory authorities.

4.3 Mechanisms for securing long term land use

4.3.1 Introduction

Securing land for long-term use as offset sites remains one of the most challenging aspects of implementing offsets and a key barrier to wider uptake. There are two key concerns relating to securing land for offsetting:

- **Gaining access to or securing land for offsetting**. The delivery of biodiversity offsets is dependent on the ability to acquire or gain rights over sufficient areas of suitable land for long term conservation management purposes**.** This can be a challenge, particularly in countries where land is scarce. It is also dependent on appropriate institutional arrangements and access to sufficient capital.
- **Securing long-term management of land for conservation purposes**. Even where land is available for offsetting purposes, the ability to achieve no net loss is dependent on securing conservation management of the land in perpetuity, even if circumstances change. This security may be enhanced by restrictions on long term ownership and/or use of the land.

4.3.2 Securing rights to land

Importance for securing long term conservation benefits

As a first step, securing long term conservation benefits of the offset requires guaranteeing ownership or management rights over the area where the offset activities will take place.

Options for implementation

The right to use land for offsetting purposes can be achieved through land acquisition or through appropriate leasing agreements. Leasing of land may provide a cheaper alternative to land purchase, but at the same time will give less security of access as leases are not granted in perpetuity.

Very few countries, either in the EU or internationally, have formal mechanisms or specific provisions in place which would provide for land to be acquired for offsetting via expropriation; instead, this is usually left to normal market mechanisms where arrangements are made on an ad-hoc basis for land to be purchased or leased. Whilst mechanisms to guarantee the supply of land might be useful if it would otherwise be difficult to legally gain access to the land in order to use it for restoration purposes (e.g. when there is considerable pressure on limited land for conflicting land uses) there may be legal or social barriers to implementing such actions.

Another concern is how appropriate the land may be for offsetting - some regulatory systems (such as the Western Cape offsetting system in South Africa) place restrictions on access to specific land areas with low ecological development potential (usually areas with high levels of existing species richness) so as to ensure additionality of conservation benefits from the offset.

In many countries, a shortage of suitable land for habitat restoration is a key barrier to development of offsets. In some cases, it may be that the land (either as a whole or for particular habitats) is in short supply and therefore opportunities for restoration are limited. In other cases, suitable land may be abundant, but ownership and rights are barriers and it is difficult to obtain the land (either through purchasing or long-term leasing) in order to use it to restore the necessary habitat.

Land acquired for offsetting may take different forms. Mitigation banks or offset proponents may acquire degraded land with a view to increasing its conservation value to a level commensurate with that of the area being impacted (as in US Wetland Banking). Intermediaries such as habitat banks may acquire this land on a speculative basis for the provision of future offset credits. A pool of areas may be integrated into a coherent nature conservation plan on the basis of existing systemic conservation goals (as in New South Wales BioBanking). This approach may have stronger societal and ecological benefits but typically entails higher transaction and administrative costs.

Integrating local farmers in long term habitat maintenance contracts can be beneficial, and supports land acquisition for offsets. The farming sector might offer land on which habitats can be created and then maintained by the present farmers, organised as a pool or habitat bank. With this the local farmers gain long term benefits and income. Empirical evidence in Tucker *et al* (2014) demonstrates that this is an important success factor for securing long term conservation benefits and for securing the 'land acquisition' process.

Box 12 Habitat banking and land use planning in Germany

The dominant role of municipalities in administering offsets in Germany has a number of benefits for accountability and maintenance of standards, even though offset requirements vary between states. Municipalities implementing offsets are required to assess the relative risks and benefits of habitat banking in their area prior to the creation of habitat banks.

In preparation for the establishment of a new habitat bank, or during the maintenance of an existing bank, a pre-assessment of the expected damage in the area relating to the bank is needed (Spang *et al*. 2005). References at this stage of the assessment are programmes and plans of regional land use planning authorities, but also federal level land planning frameworks. Some effort is required to establish coherence with existing Natura 2000 networks.

Whilst it is not usually possible to estimate the exact damage giving rise to demand for habitat banking activities, this assessment can give a useful strategic indication of how much area will be needed in the near future and which habitats will be most affected, so suitable offset land can be acquired accordingly. In line with the precautionary principle, pre-assessment also offers the possibility to avoid time lags between impact and remediation (Bunzel et al. 2009).

Pools of offset areas where ecological improvement has been previously undertaken can be acquired for provision of offset credits. This approach ensures (in theory) that no temporary losses occur, and is often referred to as 'advanced offsetting', although care must be taken by regulators to ensure that these improvements would not have occurred in the absence of the offsetting scheme. Another concern is that conservation pools may result in a patchwork of small and fragmented ecological areas, if due regulatory oversight is not in place. Strategic engagement by regulators can be important in this case - offsets need to be local enough to ensure social acceptability by local communities, but dispersed enough to maximise ecological efficiency and connectivity of habitats.

According to Bohme *et al*. (2005), appropriate areas for offsets should:

- Have a high potential for ecological development and upgrading;
- Be secured for remediation use on the long-term;
- Have a functional coherency with the impacted site;
- Not be competing with other uses;
- Be cost-effective in implementation and maintenance, and
- Not lie within current and future development areas that are exposed to projects and interventions with adverse effects on the natural environment.

International experience

The approach adopted by New South Wales as part of its BioBanking programme is a strong example of integrating land acquisition within a strategic planning framework. In this regard, planning of offsets is considered analogous to other strategic planning decisions for municipalities and provincial government. Based on current and projected planning applications, the government tries to understand where development pressures are likely to occur then puts out a call for advanced credits, recuperating this cost over time from developers. The Regional Conservation Plan for the Lower Hunter Valley (DEC, 2006) indicates where new reserves will be and identifies regional conservation priorities that are to be the focus for future offset projects (Brownlie & Botha, 2009). This reflects increasing pressure from urbanisation on the agricultural lands and remnant native vegetation across Western Sydney and along the coastal strip of much of New South Wales (Burgin, 2011), and the requirement of the Native Vegetation Act to achieve a net gain in biodiversity from development projects.

Regulators in a number of countries appear to be shifting from a preference for defined ownership rights (i.e., ownership by the offset provider) towards a more flexible approach whilst ensuring that this status is legally recognised and properly enforced once offsets are in place - changes to land titles are a relatively swift and simple means to achieve this, and have the benefit of increasing the visibility of offset areas. Although information held within land titles is typically basic, the legal status of these sites can provide a relatively robust assurance of long-term protection where appropriate monitoring is in place.

EU experience

Legal access to land is a common problem in Sweden, where there is an abundance of available and suitable land for restoration but where it is very difficult to legally gain access to the land in order to use it for restoration purposes because of a range of planning restrictions. In one case of compensation, planning conditions required that additional land be included within the Natura 2000 network to compensate for a construction project which would damage an area within an existing Natura 2000 site. In this case, the landowners were then legally required to enter their land into the network in order for it to be restored. This approach therefore was able to ensure that the land was made available.

There is evidence that EU MS have used expropriation to secure compensatory actions for Natura 2000 sites (e.g. in relation to construction of the La Brena II dam in Spain in 2004) (ICF-GHK, 2013). France has a formal mechanism for expropriating land for biodiversity compensation, however due to problems it appears to have fallen out of favour. In France land can be acquired by Agencies for Land Development and Rural Establishment (*Sociétés d'Aménagement Foncier et d'Etablissement Rural*: SAFER agencies), given they have pre-emptive rights on land for the protection of the environment and landscape (and for other objectives). However, this situation seems to have arisen due to a need for land to be purchased for it to be used for compensation. Changes have recently been made which allow developers to contract land owners or other land-users to lease the land or manage the activity in their stead.

In Germany, poor availability of land for compensation has been a long-running problem, and is rooted in strong competition between alternative land uses (Sudol and Ambrose, 2002), although this problem has decreased with the onset of new habitat banks around the country. Areas with suitable site conditions are limited especially with regard to the compensation of impacts on rare habitats, usually resulting in the selection of sub-optimal offset areas. In these cases, the site either has to be prepared at great expense (for example, through top-soil removal) or the original offset goals need to be reduced (Tischew *et al*, 2010) – neither of which would be desirable from a perspective of ensuring no net loss. Leasing of land for a defined period of time is a common solution in Germany where intended offset areas are contiguous with privately-held land. In this case, entry into the land register is an important step to ensure management practices are maintained when ownership of these land areas changes hands.

Box 13 : Middle Havel Agency, Germany

Middle Havel Agency is a private agency that offers a range of services relating to the organisation of habitat banking, including procurement of land, long-term monitoring, administration and maintenance of the offset areas. The procurement includes the evaluation of the baseline of potential areas and the possibility for upgrade, which will be determined in the habitat bank database. The agency provides legal advice in the process of negotiation and contracting for the areas. This service is charged on the basis of common fees of estate agents. The assessments in regard to nature conservation objectives are charged as additional costs. The implementation of the measures, as well as the maintenance, is usually contracted to third parties. Normal contracts for maintenance are issued for 25 years (Ecologic, 2008).

Experience from offset implementation in German Länder points to the role of **land registers** in helping developers to identify parcels of land that would be suitable for offsetting as well as ensuring that, once implemented, conservation benefits are maintained over the long term through a combination of legal recognition (by the state government) and, often, civil society scrutiny of outcomes owing to the accessibility of site information to the wider public. It is suggested that the latter can provide a particularly efficient and cost-effective means of ensuring the protection of compensation areas against development, although the ability of civil society groups to ensure the enforcement of more technical management practices may be limited.

Box 14 : Compensation area registers, Germany

Compensation area registers are directories which hold information about compensation and offset measures, including areas where these are to be implemented, are listed, updated and maintained for access. Preferably, these should be maintained by conservation authorities at the state level but in practice records on many larger projects are maintained at the federal level. The primary purposes of registries are:

- To ensure that multiple planned projects do not make use of the same compensation and offset measure/area i.e. the offset benefit be utilised and counted twice;
- To ensure that new projects do not inadvertently impact on offset areas, which would hinder the success of the measure and maintaining its benefits over the long term;
- To facilitate functional controls; and
- To help facilitate and monitor offset payments.

One essential task in the context of area and measure pools is the assignment of compensation areas to planned projects. In addition to the selection of available and suitable compensation areas and measures, these registries can also serve as the basis for targeted, future-oriented area management processes.

Public funds for the establishment of conservation areas in Germany have declined in recent years. As a consequence, nature compensation regulation has become more important, as this has a restraining influence on spatial developments. However, in a number of states there has been a trend towards greater flexibility in the specification of compensation measures for building activities by shifting the focus from quantitative to qualitative compensation. Increasing urbanisation makes it difficult to find suitable and affordable compensation land, so the objective is to use fewer new parcels of land and to upgrade the quality of nature values in existing areas- a process supported by the availability of biotope values.

In addition, more use is made of measures where more space is given to nature whilst the land keeps its existing use, for example, agri-environmental measures in extensive agriculture. This is called ‘incorporated nature protection’ without functional change and it keeps costs low (Vader *et al*, 2007).

Evidence from Natura 2000 sites and other conservation initiatives shows that many countries prefer to enter into voluntary/contractual agreements with land owners regarding conservation management, rather than undertake expensive land purchases (Kruk *et al*, 2010).

For Natura 2000, management agreements are often put in place in order to establish the management obligations (e.g. UK), but the nature conservation objectives of the land is already secured via the statutory designation. A biodiversity offsetting system is unlikely to utilise statutory designations and is more likely to rely on market mechanisms for securing land for biodiversity offsetting purposes. Therefore it may be preferable to require other forms of more secure guarantee which can ensure long-term and in perpetuity arrangements.

Notably in France, the first habitat bank, CDC Biodiversité, has entered into a 30-year agreement with the French Ministry of Environment and Forests to manage an offset site (le site de Cossure). The agreement commits the parties, regulates conditions to sell and register biodiversity units (under strict control of the French State), and describes monitoring requirements and governance for the project. There are also agreements with the adjacent reserve, local farmers managing cattle and scientists (BBOP, 2014).

Possible EU implementation issues

Whilst in general large areas of land across the EU might benefit from restoration, evidence from a selection of EU Members States (Germany, France and the Netherlands; ICF GHK *et al*, 2013) supports the view that constraints at a local level affect the available supply of that land. In reality, all countries would be expected to experience some level of constraint on the availability of certain habitats and species for remedial actions in certain locations, although this will clearly vary. Requiring like for like compensation may limit supply further, but easing like for like requirements (e.g. allowing trading up – see exchange rules in section on metrics above) can ease the situation.

All EU MS have functioning land markets and land acquisition, leasing and management agreements are all options that are widely applied for conservation purposes in the EU. A mix of these mechanisms can be used appropriately to help overcome barriers relating to supply constraints and financial cost. It is of note that for some Eastern European countries there is not yet a clear land valuation system, which whilst not directly affecting the ability to secure land for offsetting can be the source of social injustices (Wehrmann, 2010). Expropriation of land is not identified as widely used best practice. Further, whilst most MS have legal instruments that enable expropriation by the State, it is not clear that these would permit expropriation for offsetting purposes.

Agricultural land tends to be the most widely used land type for development of offsets. In a number of EU MS, agricultural land is predominantly rented rather than owned by farmers, unlike in the US. But there are examples of offset markets developing effectively in spite of high rates of land rental, particularly in Germany, where markets have been supported by the presence of habitat banks. Despite common EU land market laws, there remain some legal and institutional arrangements that can make it difficult to use agricultural land for other uses (e.g. see Schoenherr, 2014). This could present barriers in some MS where wholesale change from agricultural land was desired but not necessarily where only ‘incorporated nature protection’ is undertaken.

All EU countries have land registers. The law typically requires that all new ownership must be registered in the land register. In EU countries where the law does not have such a requirement most properties are registered anyway within a short period of time after the transaction has concluded (European Environment Agency, 2010). As well as property ownership, in almost all EU countries, leases and sub leases also have to be registered in the land register.

Whilst the land registry systems of EU countries are generally quite advanced, there is some differentiation and legal certainty of information is sometimes hard to come by (Wallis and Allanson, 2011). This is particularly the case in EU countries which also have a land cadastre (which is map-based, providing information on property boundaries) where there may be inconsistency of entries in the cadastre and in the land registry. Further, in some eastern MS public land is not included in the registry (Wehrmann, 2010).

4.3.3 Securing land for long-term conservation purposes

Importance for securing long term conservation benefits

Once land is available for offsetting, mechanisms then need to be put in place which will secure the continued use of the land for conservation purposes over the longer term. This secures the offset land tenure against future harmful human activities due, for example, to urban development, infrastructure projects or if ownership of the land changes (e.g. if ownership is transferred, or if the lease expires). The nature of these arrangements will be informed by existing regulatory and legal institutions in different countries. In some cases, voluntary agreements appear to function well in ensuring long-term benefits. Ideally, some form of formalised agreement will hold future land owners/managers to commitments over the long term.

Options for implementation

It is likely that attaching conditions to the land itself, rather than to the provider can provide greater levels of security into the future (e.g. in case the land transfers ownership or if the provider goes bankrupt). However, conditions that are placed only on the land itself may only guarantee it use as an offset site, but not necessarily its sympathetic management to achieve particular conservation goals. There may be more scope to place specific management conditions on a provider, compared to conditions placed on the use of land. Since the maintenance of long-term conservation benefits is the ultimate focus of offset schemes a combination of mechanisms that both secure long term land use and enable appropriate management conditions to be set are likely to be the best long-term solution.

The means of securing land use may be influenced by the type of impact being offset, as well as the specific regulatory and economic situation in the country or region of implementation. The type of legal guarantee over the land will depend on the nature of the property rights and the legal representation of who owns the land. The offset can be set on public or private land. In the case of the latter, it is especially important to guarantee the status of the offset over the longer term to prevent the land from being used for other conflicting purposes in the future. Guarantees can be applied either to the land itself or specific obligations can be placed on offset providers.

A range of mechanisms can be used to put these kinds of guarantees in place. Guarantees can last for a certain period of time or for perpetuity, through the use of statutory agreements and the legal instruments of covenants or easements.

Covenants run with the land and bind any successors to meet the conditions of the covenant. Covenants typically place conditions on how land can be used and require it to be managed in certain ways for the benefit of biodiversity. They are binding on whoever owns the land so the benefit is secured even if ownership changes. These are, for instance, being considered for use in the UK. However, covenants do not always apply in perpetuity; in the UK, a covenant could be released if, through a planning decision, it was considered developable land.[21]

Easements can also be granted over land which are then reflected in land titling documents and can be used to protect the conservation site into perpetuity. Easements place conditions on land titles that dictate how land can be managed. These conditions are then tied to the land title deeds so that these conditions apply regardless of who owns the land. They have been successfully used in the US, for instance. Similarly, in Germany there have been cases where there have been requirements for commitments to be included in the land title deeds and to be entered into the Land Registry.

Land ownership can also be **transferred to nature conservation organisations**, with legal clauses included in the deeds that ensure that the land will be used for nature conservation purposes in perpetuity. Leasing arrangements combined with conservation easements have particular importance in the USA, where many conservation organisations are restricted in their ability to purchase land for policy reasons, although these arrangements are usually limited to a

[21] Defra (2013) Biodiversity offsetting in England - Green paper

defined timescale. For conservation covenants or easements an appropriate 'benefit holder' must be identified, which could be either the State, a conservation NGO or the private sector. Notably for-profit companies do not hold US conservation easements (Korngold, 2011) and are not proposed to be holders of the UK conservation covenants (Law Commission, 2014). Environmental NGOs are potentially good stewards as their interests are the conservation of the land.

Alternatively, land can be **legally designated** for nature conservation purposes and thereby subsumed into a protected area network which can also secure benefits (or at least limit changes in use) in the long term.

Contracts with landholders (e.g. Payments for Ecosystem Services) are emerging as an alternative to land purchase or land registry and relate to the safeguard of specific services through management practices. However, there are no examples in practice of PES schemes being defined in perpetuity - the recent UK Law Society Consultation on Conservation Covenants, for example, highlighted the shortcomings of established PES mechanisms such as Higher Level Stewardship agreements in the context of long-term benefits. The voluntary nature of these agreements means that maintenance of management practices cannot be guaranteed with changes in land ownership over time. Nonetheless, such contracts may have some benefit within strategic landscape planning, where these are linked to established easements. Contracts could provide a 'stepping stone' to more robust conservation covenants or land registry options, but cannot represent a robust guarantee of long term conservation use.

Land registers play an important role in documenting sites that serve as offsets. They therefore help to provide a barrier to potential changes in land use or management. In some cases (such as Germany), entry of offset lands into the national or state land register is often a contractual requirement for allowing the offset to proceed. Although the information held within such registries is typically basic, this provides an added level of visibility for existing offsets and aids the informal scrutiny of offset measures by conservation groups and other civil society groups. Similarly, centrally-held registers of offset credits can aid understanding of the long-term performance of offset providers and avoid double-selling of land areas.

Depending on the context, management tools may blend aspects of these mechanisms. Terminology is also important - although easements (US) and conservation covenants (internationally) are often regarded as legally equivalent, their application and focus may differ depending on the specific legal or regulatory contexts in question. In general, a distinction can be drawn between two different types of agreement. 'Core permanent' agreements secure land from future threats such as development or changes in agricultural use (e.g. ploughing, application of herbicides, etc.), which have the potential to undo investments in conservation on the site. Secondly, complementary shorter-term agreements can secure the periodic land management needed to sustain biodiversity interests and prevent reversion to scrub or other low-value habitat (e.g. grazing, UK Law Commission, 2013).

For example, most US easements focus on restricting the use of land for development, and to a lesser extent, land management approaches that would be necessary to secure the long-term viability of conservation benefits. This reflects the specific legal and regulatory challenges facing conservation areas in the USA - particularly loss of coastal wetlands and ecosystems to urbanisation. In other settings (notably Australia and Germany) conservation covenants are seen as a supplement to land management or agri-environmental agreements.

Conservation covenants are often claimed to raise the protection of long-term conservation benefits on private land to a level comparable to protected area networks. Nonetheless, these agreements may be subject to subsequent pressure for revision or amendment. Development pressure on established protected area networks is mirrored to some degree within conservation agreements on private land. There is an increasing trend towards modification of existing easements or covenants agreements in North America, for example, where landowners who instigate these agreements seek to maximise the economic value of land or new landowners inheriting historical agreements are increasingly reported to be requesting alterations to easements or covenants. In some cases, this may indicate that further financial incentives (such as tax relief) may be needed to secure the long-term viability of such agreements. Stakeholders in the USA often suggest that greater regulatory oversight of easements and their management (e.g. by third-parties) may be necessary as government enforcement varies significantly between regions.

International experience

Easements are widely practiced in US Wetland and Conservation Banking and can provide an efficient means to secure land over the long-term, by focusing on specific land management practices together with enforcement of property rights.

The increased use of covenant mechanisms internationally stems in part from the inadequacy of funding and commitments to establish new protected areas, and a general trend in declining ecological condition of protected area networks, even as overall areas of protected land increase (Jenkins and Joppa, 2009). Covenants are seen as a targeted and cost-effective means of addressing the management of conservation on both public and private land, although their effectiveness in securing long-term conservation benefits may be limited to the extent to which these agreements can be subsequently revised or revoked.

As of 2010, approximately 30,280km^2 of land has been entered into covenant agreements in in Australia, with the average protected area around 4.1km^2 in size (Adams and Moon, 2013). Australian states typically have multiple tax incentives (from municipal to state-level) to promote the entry of land into registries and often targeted support in the production of Covenant Management Plans by local regulators. There are also a high number of voluntary covenant agreements and strategic uses of covenants by regulators with protected areas, such as to enhance the effectiveness of ecological corridors.

Discussions with stakeholders indicate that, in practice, alterations to the land registry are viewed as a quicker and more cost-effective solution to securing land over the long term in Australia, particularly as covenants are vulnerable to the legal priority afforded to earth resources over topsoil protection. Entry of land into the registry also has the benefit of public dissemination of these offset areas, although centrally-held data only relates to the latitude and longitude of offset sites, not more detailed data relating to land cover or habitat type.

The use of central databases or geographical information systems (GIS) have significant promise as a means to increase the speed and cost-effectiveness of offset proposals, as well as scientific accuracy and public transparency. For offset sites to be maintained in perpetuity, it is clearly essential that some form of record-keeping is maintained as to when and where offsets are being implemented. In New South Wales, such mapping is undertaken as part of the registry of land within the BioBanking initiative.

4.3.3.1 EU experience

A review of current practices on biodiversity offsetting across the EU (Morandeau and Vilayscak, 2012) found that the length of project developers' offset commitment in rarely determined by a regulation, with the exception of some compensation bank systems, and often correspondents to the development project's operating period. However the mechanisms employed and the average length of offset commitments varies markedly within and across MS.

In the UK, the use of conservation covenants[22] is being actively explored in the context of the government's biodiversity offsetting policy. These are voluntary agreements between a landholder and a conservation body (such as the National Trust) that seek to protect or enhance aspects of landscape value or biodiversity by permanently limiting the use of the land in order to protect its environmental condition.

[22] Conservation covenants are considered the legal equivalent of conservation easements by the UK Law Society, although their application may differ owing to the distinct legal landscape of US States

In England and Wales, there are a number of instruments available to secure biodiversity benefits through voluntary agreements (including Stewardship and Section 106 agreements) but these are typically limited in timescale (5-10 years) and geographical scope (Local Authority or county boundaries. Covenants have the added benefit of securing management requirements in perpetuity alongside registration of the land for environmental purposes, in addition to lowering costs and accelerating planning approval by simplifying legal processes.

In France, there are also plans to introduce covenants into national law. In addition, the transfer of both land and offset management to civil society organisations and public bodies is a common practice, and is seen to demonstrate commitment to ensuring long-term conservation benefits. Non-profit foundations have been established to create and manage offsets, but this has been reported to create legal difficulties where there is a discrepancy between the provision of offset land by these entities and their work in providing for-profit support and consulting services.

The designation of land has been applied in Sweden where planning conditions required that additional land be included within the Natura 2000 network to compensate for a construction project which would damage an area within an existing Natura 2000 site. In this case, the landowners were then legally required to enter their land into the network in order for it to be restored (ICF GHK, 2013).

In Spain, the national land registry has been in existence for over two centuries and plays a major role within land acquisition and rural development processes. The registry holds significant information relating to the environmental characteristics of land areas (owing to the importance of water rights and other environmental considerations for development).The registry is reportedly playing a major role in the development of formal frameworks for offsetting in Spain.

Possible EU implementation issues

Whilst **easements and covenants** exist in land and property law in the EU, in their current form they are not appropriate for broad use in a biodiversity offsetting system i.e. to act as a 'conservation' covenant or easement (Korngold, 2007 and 2011; Morandeau and Vilayscak, 2012). Two key hurdles[23] are briefly summarised here:

1. Standard easements and covenants in the EU currently need to be specified in relation to adjoining pieces of land. That is, in order to apply an easement/covenant to land for offset purposes, the benefit would need to accrue to an adjoining piece of land. This clearly restricts their usefulness as a means for securing appropriate use and management of land for offsetting as it would require offset sites to adjoin the impacted sites. A 'conservation' covenant/easement would need to overcome this, ideally allowing the benefit to be able to be held by a legal organisation that is not in possession of land that shares a boundary with the offset site (as is the case for conservation easements in the US).
2. Easements and covenants in the EU cannot generally create 'positive obligations' on the burdened party, instead focusing on 'negative obligations' (i.e. actions that can't be undertaken). A conservation covenant/easement would ideally enable positive obligations to be specified i.e. that the site must be managed for biodiversity offset purposes.

As such it is expected that the majority of MS would need to make legal provisions that enable 'conservation' easements or covenants, if these were to be utilised in order to secure the long term use of offset land for conservation purposes. There are examples of such provisions being made for other purposes in order to overcome these hurdles (e.g. statutory covenants in England and Wales (Defra, 2013)), and as discussed above the UK and France are both currently exploring the use and establishment of conservation covenants.

In many EU countries **environmental NGOs** that are independent of any state authorities own and manage land for conservation purposes. Examples include Natuurmonumenten in the Netherlands, the Wildlife Trusts in the UK, and BirdLife partners in many Member States. In those

[23] See Korngold (2007 and 2011) for a fuller discussion

Member States with strong and stable conservation NGOs, these organisations could play an important role in the provision of offsets. Assigning management responsibilities for offsets to NGOs or public sector conservation agencies could play an important role in enhancing confidence regarding the durability of land use and conservation management in the long term.

All EU MS have experience of **land stewardship** via agri-environment programmes. The major focus of these agreements is biodiversity enhancement and there is experience of engaging landowners in the design of these measures. Such agreements are used extensively in some MS (e.g. Spain, where they are of particular importance for Natura 2000 because of the concentration of land in private hands (European Commission, 2011)). However negotiating management agreements is not a straightforward process and administrative issues and high entry costs have been shown to hinder their implementation in some MS (e.g. Poland; Piazzi, 2012). Furthermore, most management agreements in the EU have been of short or medium term duration, raising questions about their suitability for ensuring long term conservation management.

4.4 Mechanisms for financial sustainability

4.4.1 Introduction

In terms of financial resources, there are three key elements to consider in order to ensure that conservation benefits are secured over the long term:

- Ensuring that there is **sufficient capital** available at the beginning of the offset to deliver the required conservation activities over time and to satisfactory standards (e.g. through the use of trust funds);
- Ensuring that there are **appropriate safeguards** if there are unforeseen needs or circumstances unexpectedly change (e.g. contingency funds); and,
- Ensuring that there are **guarantees** against risk of financial failure (e.g. bankruptcy).

4.4.2 Ensuring sufficiency of capital

Importance for securing long term conservation benefits

Ensuring access to sufficient financial capital for the lifetime of the offset is a prerequisite for securing long-term conservation benefits, since all offsets will require resources for ongoing management and monitoring. If these resources were to run out, this would at best make it difficult to ensure that the offset was meeting its objectives and delivering no net loss. More seriously, a cessation of the necessary management activities could lead to a failure to meet conservation objectives, and hence a failure to achieve no net loss.

Options for implementation

Provision of finance for ongoing management needs can be achieved through conservation trust funds - long-term funding mechanisms that are legally restricted to specific purposes and can be used to make payments to offset providers over the long term in order to meet management costs or where the trust fund provides a source of income to manage the offset into the future. The establishment of this type of fund requires that specific governance arrangements and a legal framework are in place.

Funds can take different forms and can include:

- **Endowment funds,** which invest their capital and use only the income from those investments to finance conservation activities;
- **Sinking funds**, which decline over time by using capital, as well as income, to fund conservation activity; and
- **Revolving funds**, which are supplemented by regular receipts of new resources that can replenish or augment the original capital of the fund.

Only endowment and revolving funds are able to finance ongoing activities in perpetuity, although sinking funds can be used to finance a given phase of an offset project (e.g. the site restoration phase).

Securing long-term financing for offsets is a major challenge in many countries, owing to the significant capital and management costs required to design, implement and deliver offsets over time; returns on investment may be uncertain for landowners in the absence of relevant institutions such as habitat banks. Moreover, monitoring and evaluation costs may be a significant element of overall project costs. International best practice indicates that ideally 5-10% of overall project spend should be allocated to monitoring (Lindenmayer, 2012; Franklin *et al*. 1998).

Existing evidence from the EU points to capital costs of between €30,000-€100,000/hectare for offset schemes, although actual project costs may be higher. The scale of lifetime costs may depend on the institutional arrangements in place; in Germany, costs largely pertain to land acquisition and capital costs to comply with mandated measures, whilst in the USA, costs can vary significantly depending on the acquisition value of the land.

Another key concern relating to no net loss claims is the extent to which regulators require an ‘anticipatory approach’ to impacts - that is, requiring offset proponents to secure offset benefits before provision of credits (as in Natura 2000 compensation). This can entail substantial up-front costs if there is no ability to raise funds prior to implementation of the offset. Offset planning, establishment, management and operating costs must be borne for perhaps years before specified performance standards can be applied as compensation for impacts (McKenney & Kiesecker, 2010). This approach can threaten both the growth of the offset market and the viability of mitigation banks in general. As a result, regulators in the USA and Germany usually take a flexible approach to the management of residual impacts through advance credit release or area multipliers.

International experience

Public sector trust funds have been successfully used in New South Wales in Australia, where part of the offset purchase fee is paid into a BioBanking Trust Fund which releases payments to owners of land being used for offsetting as they implement agreed activities. In the US, wetland banking schemes require bank owners to finance a trust fund with the proceeds from selling credits, which must be large enough so that the management costs can be met from interest without needing to access the capital invested in the fund.

As such, investments using endowments need to both minimise investment risk and maximise interest rates over the long term as a means to ensure the endowment remains a ‘non-dwindling’ resource over the long term (Carroll *et al*, 2008). The State of Florida, for example, requires security amounts to be adjusted annually to compensate for inflation and allow for the reality of price and currency fluctuations (Stano, 2013).

Under the new US Mitigation Rules, all compensatory mitigation projects and banks must have a ‘long-term funding mechanism’ in place to ensure compliance with long-term protection goals, supported by annual reporting of financial assurances and account balances. Whenever a long-term offset project is proposed on an area of private land, the long-term funds for monitoring compliance need to be identified and approved but *do not need to be in place prior to development*. Studies suggest as many as 90% of mitigation projects benefit from advance access to credits, which normally require land acquisition and physical/biological improvements to occur within 3 years of credit release.

Moreover, in the case of compensatory mitigation sites owned by public entities, a formal commitment from the public authority (such as a land stewardship agreement) can suffice in place of an identified financing mechanism for long-term ecological monitoring. As a result, acquired lands are often turned over to public authorities without any long-term stewardship funding arrangements- or funds pooled at the organisational level and thus vulnerable to changes in public budgets over time (Teresa, 2009).

Endowments can provide long-term assurance of funding for offset monitoring and enforcement, but ultimately represent a dwindling resource over time if the effects of inflation are not considered. The frequency of such requirements in a handful of US states such as California and Florida attests to the relative maturity of offset markets in these states and the sophistication regulatory capacities, in comparison to less-developed markets enforcing identical federal policies.

In many cases, specialised tools or guidance have been developed to support long-term stewardship goals. In California, for example, the Property Analysis Record (PAR) has been developed by the Centre for Natural Lands Management to assist landowners in calculating the costs of short-term and long-term land management based on a specific project. The tool helps pinpoint individual management tasks as well as administrative costs to calculate the overall costs of any land management project. The PAR generates an output report based on long-term funding projections including endowment fees, special district fees and other sources[24]. Such projections need to be included within initial budgeting considerations to ensure appropriate resources are in place over the long term.

Capacity is a key issue in financial budgeting and monitoring of projects. The US Compensatory Rule suggests that district engineers '*should consider'* the need to make inflationary adjustments and certain financial assumptions. It also suggests that they *'may conside*r' methods such as discount rates. Nonetheless, there are practical issues relating to the use of discounting in the design of mitigation projects - many Corps of Engineers assessors lacking the knowledge or access to financial managers to understand total return assumptions or capitalisation rates (Teresa, 2009). Experts in the USA point to examples of cost analyses alone running to 56 pages to fully meet the technical requirements of the Compensatory Rule- although many of these costs are merely 'best guess' projections[25]. Similarly, offset proponents, and particularly, land trusts engaged in the management of offsets for conservation purposes often lack the necessary financial awareness to understand their long-term liabilities and to build long-term financial plans.

While endowments are a popular method of financing habitat banking in the US, the prospective habitat banker cannot usually provide funding for the long-term management endowment upfront. This is usually resolved through an escrow approach, where an agreed share of each credit sale is diverted into the endowment account. However, the length of time it takes to fully fund the account affects the level of escrow that will be needed. Just a few years delay may more than double the amount needed spending on inflation, interest rates and other economic factors (Caroll *et al*. 2009).

Increasingly, endowment requirements relate to some form of ecological assessment such as an ecosystem or biotope valuation. In the case of rare or highly-valued species or ecosystems, this can lead to extremely high assurance requirements - one recent example from California involved a $15.8m endowment requirement to protect 2000 acres of floodplain.

EU experience

Examples of funds established for biodiversity offsetting purposes can be seen in Germany, France and the UK.

Germany: To implement offsetting before the impact and guarantee its sustainability, compensation agencies (mostly municipalities) can establish a land heritage (forests no longer used, fallow land, ponds, etc.), secure it and make it available to project developers to implement offsets. For its part, the project developer can either sign a contract for an agency to manage a land pool as part of the compensatory measures, or assess the ecological cost of its impact by calculating the number of eco-points to offset, which it will buy from the agency managing the Ökokonto. In the latter case, the sums paid to the managing agency enable it to maintain its land

[24] Centre for Natural Landscape Management
http://www.cnlm.org/cms/index.php?option=com_content&task=view&id=21&Itemid=155

[25] Personal communication, Sherry Teresa (25/06/2014).

capital or acquire new land (Morandeau and Vilaysack, 2012). In essence this provides income in the form of a revolving fund, enabling the necessary capital to be maintained.

In **France,** under the framework of pilot compensation banks, the agreement with the Ministry of Sustainable Development requires a minimum management period of 30 years and, beyond this, a guarantee concerning the ecological function of the site. Thus, if the operator of the bank is the owner of the site, it can transfer it to a perennial structure fulfilling the general-interest missions of biodiversity conservation, or associations with an endowment fund (e.g. Conservatoires d'espaces naturels – bodies responsible for the preservation of natural areas). If the land is transferred before the end of the bank's commitment period, it assigns a budget to the structure, which allows it to finance the ecological management measures (CGSD, 2012).

CDC Biodiversité financed the acquisition of 357 hectares of the Coussouls de Crau offset site orchard and became its owner in September 2008 (with The PACA land agency SAFER acting as an intermediary) (Morandeau and Vilaysack, 2012). The ongoing management operations are financed by CDC Biodiversité and reimbursed through selling biodiversity credits to developers (CDC Biodiversité, 2009). CDC Biodiversité is subsidiary of the Caisse des Dépôts et Consignations (CDC), a 190 year old French group comprising a public institution and private subsidiaries.

In **the UK**, options are currently being explored for funding biodiversity offsetting. The options being considered include (i) requiring offset providers to put in place a financial instrument such as an annuity or trust fund that will provide a source of income to manage offsets in the long term; (ii) a public sector trust fund administered by the state that would make payments to offset providers over the long term to meet management costs (Defra, 2013).

It is important to add that the capital requirements for offsets are in many cases modest in comparison to the costs of initial developments. Figures on the application of impact mitigation regulation on German highway construction, for example, assume that offset measures will account for an average of 5.4% of total construction costs (Albrecht *et al*. 2014 quoting a source of TU Berlin), although authorities may additionally request securitisation up to the value of the offset.

The municipalities that provide financial support to the majority of German conservation banks appear to be absorbing many of the long-term management costs for these offset areas. In Germany, a review published in 2005 (Böhme *et al* 2005) found that only around a third of compensation land pool administrators were passing on the costs of permanently safeguarding offset land to the project developers.

This is particularly the case with regard to strategic planning of offsets, which may result in higher net value from the offset but also impose higher direct and administrative costs for these offset credits. Developers will naturally seek offsets with the lowest costs rather than the highest benefits, and many state compensation agencies experience difficulty in selling their offset credits to developers. According to the Saxony Compensation Agency, only 30% of their available credits have been purchased because their offset credits are more expensive than others (Tucker *et al*, 2014).

The degree of public financial support to offsetting in Germany raises concerns regarding the long-term sustainability of these delivery models, in a context of constrained public investment in conservation activities amongst EU Member States. It could also raise concerns regarding additionality of offset benefits, if funds to support monitoring and administration are redirected from environmental spending elsewhere. This contrasts with the model of BioBanking in NSW, where the costs of administering offsets are incorporated within the cost of offset credits.

A distinction between 'German' and 'Anglo-Saxon' models of offset financing and delivery is the relative attribution of risk: in Germany, much of the up-front financial risk of acquiring land is borne by the public sector (specifically, municipalities), whilst in North America and to a lesser extent Australia, speculative lending of credits (i.e. before a bank reaches maturity and full capitalisation) has been seen as a pivotal factor in providing habitat banks with sufficient capital to realise offset schemes. This relative attribution of risk allows for a greater use of anticipatory approaches to project impacts under the German offsetting system, at the possible expense of flexibility and market development potential.

Alternatively, the state could establish a public sector trust fund that would make payments to offset providers over the long term to meet management costs, as in the case of the BioBanking Trust Fund in New South Wales. This fund would then be replenished through capital fees from developers. One disadvantage of this approach is that it will add to the direct costs of offsets and may constrain the development of the market (as has been observed in NSW, where overall demand has been relatively low).

Possible EU implementation issues

There are no fundamental legal issues regarding the establishment of conservation trust funds in EU MS, although the legal system is likely to influence the legal structure of a fund.

Traditionally environmental trust funds have been capitalised by accessing public sector funds e.g. from national governments or multilateral organisations. Recently, efforts been made to start to bring in increasing levels of private finance to address the increasing scale of financing needs, particularly for the large up-front costs associated with habitat banking.

4.4.3 Safeguarding against financial risks

Importance for securing long term conservation benefits

Given the long-term focus on offset schemes, there is a clear need to 'build in' appropriate safeguards in case of unforeseen needs or changing circumstances that may impact on the long-term performance of the project. These safeguards have a dual function in incentivising effective delivery of project milestones, whilst providing regulators some degree of insurance against risks of financial failure (e.g. bankruptcy of the offset provider requiring the regulator to complete the project). Safeguards are thus intended to provide a degree of ecological assurance whilst providing essential financial assurance to society.

One recent high profile example from Scotland, involving a European protected site and Scottish Coal, highlights the dangers: Scottish Coal and ATH Resources were both granted consents to mine, on the condition that they restore the site after use, only for both subsequently to declare bankruptcy. Scottish Coal alone has restoration liabilities of around £73m that are not being covered by their liquidators, after the bonds that were supposed to secure restoration were no longer equal in value to the cost of restoration. The collapse of the two major mining companies has left thousands of hectares of excavated and polluted land across central Scotland, some of which is within protected areas for wildlife (RSPB, 2013).

Options for implementation

Different financial mechanisms can be used to secure the outcomes of biodiversity conservation measures by seeking to address areas of uncertainty or risks of failure. For instance:

- **Performance bonds** are a way of securing compliance with requirements set out in the terms and conditions of an offset. The credit producer purchases a bond from the institution providing collateral, underwriting the risk of failure to deliver on mitigation or offset obligations whilst providing an incentive to do so. They can be collected in the form of cash or bank guarantee at the time of approval of the permit. Once a period of time has elapsed and performance deemed satisfactory, the bond is released. Typically, bonds are released in stages as project milestones are reached. In case of non-compliance, regulatory authorities can ensure that the bond is used to rectify the situation so that the initial obligations are met. The bond can also be used to finance remediation of unexpected effects affecting the offset.
- **Contingency funds** are intended to ensure implementation of necessary corrective actions in the event that a project does not achieve its specified goals and objectives. A dedicated contingency plan will normally identify what funds will be available for planning, implementing and monitoring any contingency procedures that will be necessary to achieve the intended

outcomes. Legal funds represent another form of contingency fund, and access to pooled funds can be of fundamental importance to small land trusts or conservation agencies that lack their own resources to respond to legal challenges.

- **Insurance schemes** may offer scope for offset providers to take out insurance to hedge against the risk of failure to deliver the necessary biodiversity benefits. The insurance could then act as a source of funds which can then be re-invested into the offset to address any areas of failure, thus allowing the offset provider to potentially still meet their obligation. Insurance can be taken out to account for uncertainties regarding whether the offset will be successful (e.g. before the offset is delivered), as well whether offset gains can be sustained (e.g. after the offset is delivered). Insurance premiums paid by offset providers may vary and could reflect the type of habitat creation/restoration scheme being undertaken, and therefore specific risks of failure (Defra 2011; 2013) There is some experience of insurance schemes being used for environmental projects in the EU. For instance, since the Environmental Liability Directive (ELD) was enacted in 2004, some insurance products are being developed or improved to cover the requirement of the ELD- the use of insurance pools may have particular relevance for offsets. .

It is possible that offset providers in the future could be required to pay into an insurance pool, which could be used to cover the risk of owners of land being used for offsets getting into financial difficulty and therefore failing to meet their obligations to create or maintain the offset (Defra, 2013)

The US Federal Guidance on Wetland Mitigation Banking recognises the following types of financial insurance for mitigation models, in addition to mandatory performance bonds (ELI, 2002):

- Escrow accounts. A fixed sum of money is placed within a third party bank account, to be held until performance standards are met- the amount required can be diminished if specific milestones or performance standards are met.
- Letters of credit. An assumption of payment responsibility by a bank or other person made at the request of the bank sponsor, usually in the form of a formal agreement or statement that the bank is required to honour.
- Irrevocable trust. A trust which may not be revoked after its creation, as in the case of a deposit of money by one organization (surety) in the name of another (credit provider) as trustee for the benefit of a third person (i.e., the permittee).
- Casualty insurance. Assurance is based on estimated costs relating to injuries or legal liability relating to project failure. A number of authorities also require assurances from extractive sectors that are based on estimated labour costs for project completion by the regulator, in case of non-compliance.

Alternative arrangements, such as corporate guarantees or self-bonds, are generally perceived as high risk and should normally be avoided by regulators (Stano *et al*. 2013).

One important consideration is that assurance safeguards (along with multipliers) are intended to address elements of financial uncertainty and risk inherent in an offset project. As with management priorities, these uncertainties may change in the course of the project lifecycle, as better monitoring or statistical processes are developed or new liabilities are discovered at the site. In this case, there should be scope to review the amount of assurance required to reflect changing project parameters (i.e. as part of an adaptive management framework). There is some precedent for this relating to securitisation against completion of remediation of mining projects, particularly in Canadian regulation. Mining policy experts (e.g. Warhust and Noronha, 2000) recommend that there should be scope to periodically review the amount of securitisation required over the life of a project, specifically when:

- New technologies or management approaches that minimise pollution are developed;

- Improvements in modelling or prediction highlight environmental risks not included in the original assessment;
- Valuable ecological, cultural or historical resources are discovered at the site; or
- The financial status of the project proponent changes.

It should also be noted that the risks of offset failure may be addressed by other means, such as metrics and multipliers to address risk and uncertainty (see Section 3). These represent a form of insurance, by increasing the extent of offsets required by a margin that allows for the possibility that a proportion of offsets will fail to meet their objectives. It could be argued that, if multipliers are applied to reflect this risk, the need for additional financial insurance may be reduced. This may explain why financial insurance is often not required for offsets internationally.

International experience

In the USA, regulators increasingly require provision of financial assurance for offsets. The three most common forms of financial assurance are letters of credit, cash escrows (money held by third parties on behalf of contracting parties) and surety bonds (Kett, 2013). Whilst there is some evidence that these forms of assurance incentivise greater compliance with performance criteria in completed projects, stakeholders indicate that these assurances may make offset markets increasingly prohibitive to new market entrants.

The main distinction between different forms assurance is the relative allocation of responsibilities, liabilities and financial risk between the different parties. For this reason, a combination of these forms of assurance is commonly used as a means to pool project risk, particularly in the USA and Australia (NSW EPA, 2002), so as to provide incentives to all parties to deliver on their contractual obligations. The ICMM (2005) in particular has highlighted that spreading the cost of offset development can reduce the overall risk of project failure. Another concern is the extent to which assurance measures address long-term risks arising from the offset project. Although benefits should be realised so as to be provided in perpetuity from an offset, most forms of assurance (with the exception of endowments) have a defined period of implementation. One study of mitigation banks (ELI, 2002) found that, whilst the majority of banks require financial assurance (usually in the form of performance bonds) and information relating to long-term management of the site, few hold information relating to long-term financial management of the site.

Also in the US, a 15 – 25% contingency fund is normally set aside for additional work in case a project fails to deliver or experiences unforeseen costs. These funds are intended to address corrective measures, which should be outlined within a contingency plan submitted ahead of credit release. This fund should normally be separate from any other financial assurance. Legal defence funds have assumed increasing importance in the USA with the tightening of monitoring and enforcement processes arising from the Compensatory Rule and other regulatory developments. In some cases, lawsuits impose costs in region of $1 million per case. However, many land trusts engaged in the ownership or management of easements have no dedicated resources for legal defence (and often a limited awareness of their potential liabilities), so a legal challenge may lead to the collapse of the institution and place the status of the offset in jeopardy. To remedy this, national umbrella associations such as the Land Trust Alliance have developed in-house pools of conservation easement insurance intended to address such eventualities, and require contributions to a legal defence fund (i.e., from endowments) within their standardised budgeting procedures.

According to US stakeholders consulted in this study, a consequence of the Compensatory Rule for wetland mitigation has been a significant growth in the use of security products so as to address a range of potential liabilities arising from each compensation project. Often, the level of securitisation required is a reflection of the scarcity value of the habitat or species concerned- and in some cases these security requirements can be prohibitively expensive for developers.

In Australia, New South Wales Department of Primary Industries, which operates a policy of no net loss for marine developments, requires a monetary bond to insure against the risk of project failure (Burgin, 2008). The scale of the bond required will typically relate to a quantitative or monetary estimate of scarcity value of the habitat or species involved - in 2002, up to $250,000 was required as a bond for seagrass habitat (NSW EPA, 2002). Similar examples of scarcity values being applied to assurance requirements have been seen in California- one recent project involved an endowment requirement of $15.8m to offsets for 2000 hectares of floodplain mitigation[26].

EU experience

In Germany, authorities have the power under the Federal Nature Conservation Act to request a security up to the value of the offset.

In England, one of the options suggested by the recent Green Paper on Biodiversity Offsetting is that offset providers could be required to pay into an insurance pool that would cover the risk that the owners of offset sites get into financial difficulty and fail to meet their obligation to create or maintain the offset site.

Although there is limited experience in the EU in applying safeguards in relation to offsets, there are precedents under other legislative frameworks and requirements. For instance, the EU legislative framework on mining waste addresses the financial implications of environmental damage resulting from accidental pollution or other damage. The Mining Waste Directive (2006/21/EC) provides for “financial guarantee, in the form of a financial deposit, before operations that involve the deposition of extractive waste may begin. The financial guarantee has to cover obligations under the permit and the rehabilitation of land affected by the waste facility.”

Similar requirements are in place within some Member States. Mining regulations in Sweden, for example, require mandatory posting of securities as part of the site environmental management plan under the Environmental Code. Around 20% of abandoned contaminated sites in Sweden are attributed to the metals sector, and rehabilitation of abandoned mines is often a precondition for companies developing new mines. The Environmental Code includes an ordinance requiring payment to a government-administered environmental damage insurance fund where no liable party is available (CCSG, 2001).

Various other directives have mandatory financial security to ensure that operators have adequate funding to pay for known environmental liabilities such as closing a landfill (Box 10).

Box 10: EU Directives requiring financial guarantees

The Council Directive on the landfill of waste (1999/31/EC), the Council Directive on high-activity sealed radioactive sources and orphan sources equipment (2003/122/Euratom), the Directive of the European Parliament and of the Council on waste electrical and electronic equipment (2006/96/EC), the Regulation (EC) N°1013/2006 on shipments of waste all require a person who ships waste to have a financial guarantee in the event the shipment or recovery of the waste cannot be carried out as intended or the disposal is illegal. In addition, the Directive on the geological storage of carbon dioxide requires proof of “financial security or any other equivalent, on the basis of arrangements to be decided by the Member States”. The financial security must cover obligations under the permit, including closure and aftercare provisions or any obligation arising from inclusion of the storage site under the EU Emissions Trading Scheme.

Another legal framework which provides a basis for financial security is the Environmental Liability Directive (2004/35/CE). Article 14 of the Directive states that national regulations should encourage the development of financial securities or, of equivalent measures, to enable operators to use guarantees to cope with their environmental liability, regarding species and natural habitats protected by the Birds Directive (2009/14/EC) and the Habitats Directive (92/43/EEC), the Water Framework Directive and the direct or indirect

[26] Personal communication, Sherry Teresa (18/05/2014).

contamination of land incurring with significant risk on human health. Since the Directive was implemented, several Member States have included mandatory requirements for providing financial security. This is the case of Spain, Bulgaria, Czech Republic, Hungary and Slovakia. The main reason is the high costs incurred by public authorities in the restoration of "orphaned" polluted sites or in case of insolvency of the operators.

Also with respect to environmental liabilities, some countries have promoted the use of bank guarantees (Australia, Belgium, Cyprus, Czech Republic, the Netherlands, Poland, Spain, and UK), or insurance pools (Spain, France and Italy) or other market mechanisms such as funds or bonds (Austria, Belgium, Cyprus, Poland and Spain). Different types of insurance schemes have emerged as standalone products to respond to specific requirements of the environmental liabilities or have elevated the existing products to adapt to individual liabilities needs.

Source: BIO (2012). Implementation Effectiveness of the Environmental Liability Directive (ELD) and related Financial Security Issues

Possible EU implementation issues

Within the EU, there is growing experience of environmental liability insurance in the context of the Environmental Liability Directive, whilst markets for environmental liability insurance products (relating to land and water damage) are particularly well-developed in a number of northern EU Member States, where this was already contained within statutory legislation or environmental codes. Insurers operating in such mature markets (where capacity typically exceeds demand) often provide security to companies operating in less-developed markets of the EU.

The European Commission is currently considering introducing a compulsory requirement for environmental insurance under the ELD. A recent survey (Insurance Europe, 2014) analysed environmental insurance products in 18 MS across the EU. One notable trend in recent years has been the continual growth of environmental insurance pools (covering both national and international markets) across the EU, alongside rapid growth in individual insurance products. Companies in Spain, France and Italy, for example, can access specialist environmental liability pools to address insurance requirements, including damage and mitigation costs, defence and other legal costs. These pools could conceivably provide ex-ante security for a range of offset measures, as a natural extension to mitigation costs.

The majority of ELD cover is between EUR 1m and EUR 5m - although available capacity in some markets is up to EUR 50m, and can be higher on request. Amounts vary according to domestic demand, but cater to the distinct needs of SMEs and large corporations in different countries. Cover is generally available for all ELD risks, including primary remediation, compensatory remediation and complementary remediation.

The scale and diversity of environmental liability insurance available across the EU would seem to indicate that the EU insurance industry is well-placed to deliver the range of specialist insurance products that provide security for mitigation projects in the USA. However, it is important to caution that overall claims on ELD related insurance products are very low, and this may influence the liquidity of the market. The more complex, long-term nature of offset projects could imply a higher probability of claims, and potentially affect the availability and cost of relevant financial services.

4.5 Conclusions on mechanisms to secure long term conservation benefits

Whilst markets for conservation offsets exist at different levels of maturity, and are influenced by very different institutional and geographical environments, a number of general findings can be drawn with regard to securing long-term conservation benefits.

While many studies and evaluations address the refinement of the theoretical and design aspects of offset projects, comparatively few address issues of practicality and feasibility in existing projects (Bull *et al*., 2013). The evidence reviewed in the course of this study and views of consulted experts point to uneven implementation and significant discrepancies in quality management for common regulatory systems. Clearly, a balance must be found between systems that are suitably prescriptive to establish common minimum standards for maintenance of long-term benefits, and systems that are realistic and achievable, as well as those that can be maintained over time.

4.5.1 Management and regulatory systems

The review indicates that certain requirements need to be in place to provide confidence that offsets will deliver long term conservation benefits. Offsets need to be underpinned by a binding contract or agreement, linked to a long term management plan, informed by effective monitoring and reporting arrangements, and backed by effective regulation and enforcement.

Offsets may be implemented either through existing regulatory frameworks or through new arrangements. Long term conservation benefits depend on the capacity of regulators to approve, regulate and enforce offset arrangements and to ensure adherence to management plans. The private sector has an important role to play in planning, delivery, monitoring and reporting; accreditation and certification of providers and consultants can contribute to this process.

4.5.2 Land use and land management systems

Long-term stewardship of land needs to be realised through transparent assignment of responsibilities relating to ownership and management. Mandating the entry of offset areas into a centrally-held registry is a swift and effective means towards increasing visibility of offsets and facilitating civil society and regulatory scrutiny. Additional mechanisms such as covenants or easements provide a more reliable means of ensuring long-term use of the land for conservation purposes but do not preclude the need for ongoing monitoring. Management agreements have some promise as an efficient means to achieve short-term management activities but do not provide a reliable mechanism for securing long-term conservation benefits. Critically, land management agreements do not address the underlying drivers of land use change and threats - which can be best mitigated through strategic management by governments and regulatory bodies.

4.5.3 Financial sustainability mechanisms

Financial sustainability of conservation benefits should be ensured by an effective guarantor with capacities to ensure the delivery of the offset over the long term, such as a trust fund. In this situation, the guarantor assumes the long-term financial responsibility for the offset in return for capital fees that cover these costs.

To ensure that the funds held by such an institution are responsibly managed and will be available for the lifetime maintenance of each offset project, accreditation processes for institutions such as conservation banks and trust funds should be put in place wherever possible. The rigorous accreditation process for US Mitigation Banks can be seen as best practice in this regard, as can the requirement of ex-ante access to capital for accreditation of French conservation banks.

The next alternative is to request security directly from the developer associated with the project, to address the specific risks and liabilities associated with the offset. This has the benefit of clearly defining financial liability from the outset, which may be desirable for companies looking to become involved in offsetting.

Ideally, financial security systems should incentivise timely completion of the project, and protect regulators from financial liabilities in the case of project failure. Linking security requirements to key project milestones (rather than requiring all related security up-front) can be a way to incentivise the responsible management of the offset whilst minimising the already large capital costs of offsets.

It is also good practice to include contingency funds within long term financial planning, in order to address unforeseen problems or expenses. Larger scale risks, such as technical failure of the offset or financial failure of the provider, should also be addressed in a proportionate manner. This may be achieved through a variety of financial guarantees and securities, or through alternative means such as offset metrics.

4.5.4 Summary of requirements

The review in sections 4.1 to 4.4 suggests that, in order to provide secure long term conservation benefits, an offset must:

- **Be based on a binding contractual agreement** – i.e. the developer or provider makes a legally binding commitment to deliver the offset; this is a condition of the permit for the development; the contract/ permit specifies certain conditions that need to be complied with (e.g. regarding management actions, monitoring, reporting, financial aspects); and the regulator has the ability to enforce these conditions. The nature of the contract may vary according to the planning/ permitting/ regulatory structure in place;
- **Involve a long term management plan** – adherence to which is likely to be a condition of the contract. This will specify required actions, performance standards and targets, monitoring and reporting arrangements;
- **Secure rights to manage the land for conservation purposes**. This is most likely to be achieved through purchase of that land, although long term leases or long term management agreements specifying conservation actions are a possibility (with the proviso that they do not offer the same levels of long term security);
- **Involve obligations to use the land for conservation purposes in the long term / safeguards against changes in use**. This may involve a covenant or easement which specifies long term use, involvement of a 3rd party such as an NGO committed to conservation use, or long term regulatory oversight / public scrutiny, perhaps backed up by information tools such as registers which specify that the land is to be used for conservation purposes;
- **Demonstrate secure access to finance** to fund conservation action. This will normally be achieved by requiring establishment of an appropriate conservation fund, though there may be alternatives (e.g. bank guarantee); and
- **Provide safeguards against risk of failure**. Such safeguards may be achieved through: metrics (e.g. the requirement for offsets includes a risk multiplier that allows for a certain % failure); regulatory measures (i.e. the regulator secures all reasonable safeguards); contingency funds (additional funds are added to allow for unforeseen costs); and/or financial insurance (insurance is provided against risk of technical or financial failure, perhaps through a collective pool into which all offset providers pay.

The following section draws on the analysis in sections 3 and 4 to examine options for metrics and mechanisms for long terms sustainability of conservation benefits, in the context of the EU No Net Loss initiative.

5 Options for Implementing Biodiversity Offsetting in the Context of the EU No Net Loss Initiative

5.1 Metrics

5.1.1 Aims, scope and approach

Building on the conclusions from the review of metrics provided in Section 3 and Annex 1, this section of the report assesses the applicability of the main approaches to measuring biodiversity and ecosystem service losses and gains through metrics to address biodiversity losses in the context of the EU no net loss initiative. The analysis primarily focusses on the stated aims of the no net loss initiative as set out in the EU Biodiversity Strategy and the possible policy options that may implement it, as described in the recent report for the Commission (Tucker *et al*, 2014). It also takes into account other biodiversity targets, in particular Target 1 on implementing the Habitats Directive (including improving the conservation status of habitats and species of Community interest) and Target 2 on ecosystem restoration and green infrastructure (including the potential contribution the EU's forthcoming restoration priority framework[27]) and wider related EU policy goals (e.g. relating to the Water Framework Directive (WFD), Marine Strategy Framework Directive (MSFD), Resource Efficiency Roadmap and relevant climate objectives). The effects of relevant Member State policies, legislation and other initiatives that aim to achieve no net loss and/or contribute to the measurement of biodiversity losses and gains are also considered. Thus the overall assessment reflects on the degree to which metrics could contribute to the measurement and thus demonstration of no net loss in relation to a variety of polices and across a range of sectors and scales (e.g. from local projects to regional, national and EU wide assessments of policy impacts).

Practical issues are taken into account in the assessment, such as the availability of data and biodiversity knowledge and expertise, and existing initiatives, such as requirements for monitoring under the Birds and Habitats Directives and wider development of biodiversity indicators under the SEBI process. In particular the analysis considers how the use of potential metrics could interact with efforts to map and assess changes in ecosystems and their services under the MAES initiative (Box 7). The analysis therefore considers the compatibility of different metrics to the approaches and indicators being used in MAES, and examines the implications for measuring residual losses and potential gains through offsetting, with a view to achieving no net loss.

The assessment is set out in Table 5.1 using the simple metric typology developed in section 3 and the following set of criteria that aim to capture the key issues of interest outlined above:

- **Potential to contribute to the EU aim of achieving no net loss** of biodiversity and ecosystem services, and the degree of certainty and robustness that metrics offer in this respect;
- **Potential to contribute to other EU Biodiversity Strategy targets and related environmental objectives** (in particular EU Biodiversity Strategy Targets 1 and 2, WFD and MSFD);
- **Ability to address the range of different impacts** on biodiversity and ecosystem services, which include land management, land use change, resource use, pollution and invasive alien species as well as built development;
- **Applicability at different levels** – including the project, organisational, regional and sectoral levels;

[27] The restoration priority framework which was under development at the time of this study, seeks to develop a common understanding of how the target to restore at least 15% of degraded ecosystems will be delivered across the EU, and a set of criteria for identifying the priority actions at national and subnational level.

- **Requirements for and compatibility with existing data and indicators** of biodiversity and ecosystem services loss (e.g. monitoring under the Birds and Habitats Directives, SEBI indicators and the MAES initiative);
- **Requirements for offset site and impact site data** (e.g. regarding the type of and condition of habitats present, species presence and abundance and relationship with habitats and the of importance of the areas for ecosystem services);
- **Reliability and transparency** – if properly carried out (e.g. by trained personnel with adequate data); and
- **Overall practicality**, taking into account data, skill, consultation and institutional support and scrutiny requirements, and associated costs.

Drawing on this analysis, the assessment then considers how these approaches might work in the wider EU context, taking account of differences amongst the Member States in legal frameworks, regulatory structures, and potential variations in skills and capacity. The assessment identifies the different conditions needed to make offsets work in practice, considers the extent to which these are present across the EU, and examine the implications for the potential different approaches to defining metrics, particularly considering the degree to which standardised approaches are appropriate and to which flexibility is needed. This then leads to the identification and development of a number of policy options for supporting the use of metrics in achieving the EU's no net loss initiative. These are then examined in terms of their overall advantages, disadvantages, applicability across the EU and potential contribution to the EU no net loss initiative.

5.1.2 Assessment of metric options

Table 5.1 Assessment of the ability of the main types of metric to contribute to the EU no net loss initiative and other environmental objectives

Metric	Potential to contribute to the EU NNL objective	Potential to contribute to other EU environmental objectives	Range of impacts addressed	Applicability at different levels	Requirements for and compatibility with existing data and indicators	Requirements for offset site and impact site data	Reliability and transparency	Practicality, including cost effectiveness
Habitat (biotope) area	Very low as the metric is only appropriate for habitats of very low biodiversity value including for species, wider use would lead to biodiversity losses.	Very low as too simplistic and doesn't capture losses and gains in condition	A crude assessment of habitat loss/change only	All, i.e. project to EU and organisational/ sectoral, if a common habitat typology is used	Variable amongst MS, basic EU data exist (CORINE) which is suitable for some habitats, but habitats of high ecological value are often inadequately mapped	Area of impacted habitat	High provided that habitat types are correctly identified and mapped	High as very simple metric
Habitat (biotope) area x standard value	Moderate as the metric can be applied to all habitat types, and it can facilitate trading up in offsets; but biodiversity losses may occur if habitats in offset sites are of lower condition than impacted sites. Poor method for species with special requirements.	Low relevance	Principally only habitat loss/change only	Project to regional or national level, depending on scale of standard valuation; and organisational/sectoral	As above for area, and information on the ecological value of habitats varies	Area of each habitat type	Moderate, most valuations are expert judgment and thus may not be replicable, consistent or transparent	Moderate/High as required information is normally available, but valuations are judgements that should be subject to consultation

Metric	Potential to contribute to the EU NNL objective	Potential to contribute to other EU environmental objectives	Range of impacts addressed	Applicability at different levels	Requirements for and compatibility with existing data and indicators	Requirements for offset site and impact site data	Reliability and transparency	Practicality, including cost effectiveness
Habitat (biotope) area x site condition	Moderate as the metric provides a more refined measure of the biodiversity at each impacted and offset site, but by itself it is only suitable for like for like offsets. As above for species.	Could contribute to restoration linked to BS Target 1 (outside N2K sites to ensure additionality) and Target 2, also WFD and MSFD	All impacts that can potentially lead to habitat loss/change and change in condition	Could be project to national and organisational/sectoral levels if common condition criteria agreed, but defining these at large scales and across borders would be problematic, hence favourable conservation status is assessed nationally	Variable data and understanding of condition depending on habitat and MS	Area and site condition (in detail for high value habitats) of each habitat type	Variable as assessment of condition can be complex and good indicators require a reliable understanding of ecological condition and benchmarks for each habitat type	Moderate / variable as requires detailed site assessments; since the alternative of simple condition assessments are likely to be unreliable and too subjective
Habitat (biotope) area x standard value x site condition	High as it explicitly and systematically considers the potential value of habitats and their actual site condition. Facilitates trading up and offsets that address habitat condition as well as type. Above for species.	Could contribute to restoration linked to BS Target 1 (outside N2K sites to ensure additionality) and Target 2, also WFD and MSFD	All impacts that can potentially lead to habitat loss/change and change in condition	Project to regional or national and organisational/sectoral depending on scale of standard valuation; no greater than national for reasons above	As above for area x site condition	Area and site condition (in detail for high value habitats) of each habitat type	Tend be complex and of variable reliability due to the difficulties of measuring condition	As above

Metric	Potential to contribute to the EU NNL objective	Potential to contribute to other EU environmental objectives	Range of impacts addressed	Applicability at different levels	Requirements for and compatibility with existing data and indicators	Requirements for offset site and impact site data	Reliability and transparency	Practicality, including cost effectiveness
Species focussed approaches	Low by themselves as they can only be applied in practice to a few species and hence a small proportion of biodiversity, but if combined with habitat measures they can avoid losses of species of high conservation value	Could contribute to restoration of habitat for species of Community interest, thus supporting BS Target 1 (outside N2K sites to ensure additionality)	Variable as it will depend on the species considered and their sensitivities	Probably project to national levels as defining species requirements (e.g. through HEP) at larger scales and across borders would be problematic	Variable and ecological requirements for many species of high conservation concern are not well known enough to be reliably quantified	Detailed and up to date data for each assessed species on their use of impacted sites and offset areas, so lengthy field surveys or good spatial records are required	Complex metrics, especially where multiple species are assessed, and assessments of habitat suitability are often expert judgements, which may vary in reliability and repeatability	Impractical for wide application due to narrow focus, high data requirements per species and associated costs
Replacement costs	Uncertain, as it is only appropriate for fee in lieu type systems, and their use will depend on policy options take up under the NNL initiative. It needs to be informed by other metrics to specify what type of habitat needs to be replaced, and ideally what its condition should be	No link to other objectives	None directly	Projects only	Reasonable data on management, restoration and creation costs are available from agri-environment schemes; costs for some habitats are uncertain	As for habitat area x metric	General habitat restoration costs are reasonably certain, but individual requirements may vary considerably	Moderate, as key data exist, but also needs to be informed by another habitat metric

Metric	Potential to contribute to the EU NNL objective	Potential to contribute to other EU environmental objectives	Range of impacts addressed	Applicability at different levels	Requirements for and compatibility with existing data and indicators	Requirements for offset site and impact site data	Reliability and transparency	Practicality, including cost effectiveness
Ecosystem service specific metrics	High if they are selected for this purpose	Potential to link to a wide variety of objectives, but would need some standardisation of service types and metrics and new approaches to combining them	Variable as it will depend on the ecosystem services considered and their sensitivities	Probably project level for most services due to lack of standard metrics for each service, but potential for some standardisation	Data being collected as part of MAES and related national ecosystem assessments etc, but are often of low relevance to project level impacts or are complete	Detailed site scientific and/or cultural value data	Variable depending on the service and the site, but generally low reliability and transparency as the quantification of ecosystem services is difficult	Variable depending on the service and the metric being used, but reliable measurement of some services is complex, time-consuming and expensive; basic mapped proxy measures are unlikely to be suitable in most cases
Ecosystem service valuation	In theory, high potential, although this is notably constrained by limitations of methodologies and restricted data availability, resulting in a low potential.	Potential to link more broadly to economic analysis of environmental policy and projects	Variable as it will depend on the ecosystem services considered and their sensitivities	Applicable at all levels given the standard monetary metric	As for ecosystem service metrics. Further, appropriate valuation data to use in assessment is very limited	As above plus information on local beneficiaries values	Low as combines problems with quantifying ecosystem services with the substantial difficulties associated with their economic valuation	As for ecosystem service metrics. Further problems include obtaining valuation evidence as existing evidence is limited and primary work likely to be prohibitively expensive

The analysis above suggests there are unlikely to be significant barriers to the use of habitat area x value metrics. Indeed the proliferation of many habitat value lists in Germany is probably an indication that their development is not particularly difficult; time consuming or expensive, although good practice should involve adequate consultations amongst biodiversity experts and other stakeholders. However, the feasibility of using some of the other metrics is more likely to vary considerably amongst EU Member States. The assessment of habitat condition is a complex process and not one that is currently carried out in depth in any EU country as part of offsetting. Although a simplified condition assessment has been piloted in England, this is widely considered to be too basic and the assessment methodology is not fit for its purpose (Environmental Audit Committee, 2013). The use of metrics that include condition would therefore require significant development in all Member States, as well as later training for authorities, developers and consultants etc in the use of the metric.

The use of species and ecosystem service metrics would result in even greater research requirements and demand for institutional support advice and training. However, information for species metrics would only be required for species of high conservation importance and data on these are available in many countries. Furthermore, the key data requirements are understood (i.e. species occurrence records, abundance estimates and habitat suitability assessments) and there are no major barriers to obtaining them. Although the collation of such data and incorporation into a GIS has significant costs, these are unlikely to be prohibitive and may even result in strategic savings through faster planning decision making.

By contrast, the development of ecosystem service indicators is still at an early stage and therefore a greater understanding is required of what strategic data may be required to support their application. Although initiatives such as MAES are investigating this issue intensively, it is likely to be many years before detailed ecosystem service metrics can be consistently applied across all EU countries.

In considering the **costs** and **cost-effectiveness** of offsetting metrics, it is important to consider both:

- The **administrative and transactions costs** of developing and applying the metrics; and
- The overall effect of the metric on the **cost of delivering the offset** itself**.**

Data on **administrative and transactions costs** relating to offsetting are lacking. The analysis is therefore largely qualitative. Overall, while the costs of metric design and implementation are typically a very small proportion of the costs of offsetting, it is clear that more complex and data intensive metrics are more costly to implement than simple ones. However, even the most complex and costly metrics are likely to be a small proportion of the costs of offsetting moderate to large biodiversity impacts. For small projects and others with very low biodiversity impacts it may be necessary to consider balancing the needs for ecological rigour and the feasibility and cost. It is also important to bear in mind that proactive investment in biodiversity and ecosystem service mapping can help reduce the costs of assessing potential impacts and gains, especially in terms of avoiding project delays and associated opportunity costs.

The **cost of delivering the offset itself** is directly affected by the extent of the offset requirement, which in turn is directly influenced by the metrics, multipliers and exchange rules that are applied. The choice of metric therefore has a direct bearing on costs, as do the values applied in its application. It is important to recognise this in the choice of indicators and scoring systems applied. For example, whether habitat A is considered to be twice or four times as valuable as habitat B can double or half the level of the offset needed, with a direct effect on the costs of offset provision. The effect of metrics on costs is especially apparent with respect to the application of multipliers to account for risk, uncertainty and/ or time preference, as these have a proportionate effect on offset requirements and therefore directly influence costs.

A study by GHK and eftec (2011) for Defra examined the costs of biodiversity offsetting in England, and demonstrated that these costs are highly sensitive to the choice of metrics and multipliers applied. Overall, the study found that, dependent on assumptions regarding different variables, the cost of a policy to offset current losses of biodiversity through development in England could be between £50 million and more than £400 million annually. The metrics employed and assumptions governing them had a large effect on the estimated costs, which were found to be sensitive to the use of risk multipliers as well as assumptions about the condition of the land that is developed.

The assessment in Table 5.1 above, and consideration of the applicability of the metrics to EU Member States, confirms that there is no one type of metric that would clearly be the most applicable to supporting the EU no net loss initiative and other related objectives. This is primarily because, as noted in the conclusions in section 3.6, each of the metrics has its own general advantages and disadvantages in terms of its efficacy and efficiency in providing reliable and accurate measures of losses and gains in biodiversity and ecosystem services. Furthermore, this is reiterated when the metrics' potential to support specific EU biodiversity and ecosystem service objectives is considered as well as their practicality in terms of utilising existing data sources, knowledge and skills. For example, whilst metrics that are based on standard valuations of habitats and assessments of actual habitat condition provide the most reliable measures, it is clear that these metrics are relatively demanding in terms of data requirements and normally require site surveys. Although data on the costs of applying each of the metrics (e.g. in terms of site surveys etc) were not found in this study, it is likely that the direct costs of ecological surveys are normally relatively low compared to other project costs. However, surveys can be time consuming (often taking a year or more in order to assess seasonal patterns). Therefore, their application to all projects (particularly to small projects with minor impacts) would probably be considered unreasonable amongst developers and authorities. Furthermore, focusing resources on the application of this type of metric would reduce the likelihood that supplementary species and ecosystem service assessments would be carried out where required.

There is therefore a requirement to balance the need for reliable ecological knowledge with the need to ensure that burdens on regulators and developers are reasonable. One way of dealing with these opposing needs is to develop an offsetting framework that uses a range of types of metric with varying requirements that are proportionate to the risks of biodiversity and ecosystem service losses that they address. Thus for example, highly artificial habitats with low ecosystem service values in low quality landscapes could be assessed with basic baseline studies and a very simple habitat x area metric, or the restoration cost metric. Projects impacting semi-natural and natural habitats or habitats within important areas (e.g. defined ecological networks) would warrant the use of more sophisticated metrics and full assessments. These metrics should be supplemented with species-focussed metrics if species of high conservation importance occur in the development impact zone that cannot be reliably assessed using general habitat proxies.

However, it is also important to bear in mind that serious problems can arise if there is a proliferation of different metrics used within a country, as for example observed in Germany (Bruns, 2007). Therefore it may be appropriate to develop national or regional legally underpinned guidelines or frameworks for metrics, which outline broad approaches but do not set out detailed methods and values.

5.1.3 Assessment of policy options for metrics

Considering the results of the above assessment of metrics it is clear that it is not appropriate to propose the use of one single type of metric in supporting the EU's no net loss initiative. However, **there is a strong case for encouraging the promotion of key principles and best practices in the development and application of biodiversity metrics (including associated multipliers and exchange rules).** This could be achieved through:

- **Establishment of common principles –** setting out minimum requirements according to key good practice principles relating to metrics (such as on appropriate metric types/currencies and measurement methods for different levels of biodiversity and impact, use of multipliers and exchange rules), whilst allowing flexibility in their detailed specification and application according to needs and constraints; and/or

- **Development of a standardised set of metrics** – the specification of a set of different metrics (e.g. in terms of type/currency and associated multipliers and exchange rules), that address the needs of different levels of biodiversity and impact in a proportionate and practical way in accordance with key best practice principles, that would be applied across the EU.

The advantages and disadvantages of each of these approaches, which could be applied separately or in combination, are summarised in Table 5.2 and further discussed below. These approaches are also compared to the existing situation whereby offsetting and the use of associated metrics follows a flexible approach with a variety of metrics and associated methods and guidelines being developed and used across the EU according to national and regional needs and decisions.

Table 5.2 Summary of the main advantages and disadvantages of approaches for supporting the use of metrics in contributing to the EU's no net loss initiative

Option	Advantages	Disadvantages
Flexible approach (i.e. the current situation)	Facilitates the use of metrics that are suited to national and regional needs, opportunities (e.g. available data) and constraints (e.g. knowledge of biodiversity and ecosystem services). Enables unrestrained innovation. No agreement necessary, so will not be a constraint on the development of a NNL initiative.	Can lead to varying standards due to the variety of metrics and principles being followed, which also causes confusion and undermines attempts to ensure consistent effective and efficient policy implementation and desired coherent and equitable policy impacts. Hinders the pooling of knowledge and sharing of data and development of supporting tools and institutions etc.
Common principles	Builds on previous experience and identifies key lessons and agreed minimum standards based on good practice, thereby facilitating the selection, development and use of effective and efficient broadly consistent metrics that are widely understood and accepted, whilst allowing innovation and flexibility to adapt to specific needs and constraints. Likely to be supported by key stakeholders and will be easier to agree on than specific metrics.	The identification and agreement on key principles, minimum standards and practice lessons will be challenging due to the varying needs and constraints across the EU and uncertainty over how the NNL initiative will develop and stakeholder support for it. The flexibility and varying interpretations and applications of it may still lead to a wide variety of metrics with some of the disadvantages of the flexible approach above.
Standardised set of metrics	Will provide the clearest and most consistent recommendations, which should (if correct) lead to the development of the most effective, efficient, replicable, comparable and widely used and understood metrics. Will best facilitate sharing of knowledge and tools and could enable combination of metric data to monitor NNL achievements at sectoral, regional, national and EU level.	Will be difficult to develop and agree given the variety of needs and constraints across the EU. May constrain the development of new innovative metrics.

On the basis of these advantages and disadvantages, set out above, it is apparent that establishing a common set of metrics would be extremely challenging technically, and difficult to negotiate and to agree upon. It is suggested that for the time being, **the most appropriate**

policy option would be to develop common principles on the development and use of metrics, perhaps accompanied by suggested important features of different metrics that could suit different situations. This would allow Member States to decide on the detailed design of their metrics according to their circumstances and established approaches, but would ensure minimum standards are consistently followed and good practices encouraged.

An outline of some of the key principles that could be further considered is included with the conclusions in Section 6.

5.2 Mechanisms

Based on the research carried out and presented in this report, this section reviews the strengths and weaknesses of the principal options through which mechanisms can be put in place to secure long term conservation benefits. It qualitatively assesses the available bundle of options for each of the three broad mechanism elements against the following criteria:

- Robustness: likely success in enabling remedial action for offset purposes;
- Durability: likely delivery of conservation benefits in perpetuity;
- Flexibility: applicability at different levels (i.e. from project to sector), locations and scales;
- Applicability: compatibility with existing EU MS legal and regulatory structures;
- Feasibility: technical or administrative capacity to deliver; and
- Cost: administrative and compliance costs.

5.2.1 Management and regulatory tools

The review above concludes that securing long term benefits depends on a binding contract, linked to a long term management plan establishing appropriate performance standards and monitoring systems, which can be legally enforced by the regulator. The form of the regulatory system adopted is likely to vary between countries, reflecting variations in existing systems of planning, development control and environmental regulation. In some Member States offset requirements may be specified in new regulations, which may set binding national rules or more general design principles which allow for more flexibility in application by local or regional regulatory authorities. In other Member States offsets will be more effectively delivered through existing regulatory arrangements, but there will be a need for guidance specifying the principles and practices needed to ensure the delivery of long term conservation benefits in order to secure no net loss.

The manner in which these options are adopted or combined with other approaches (for example, strategic conservation planning) will depend on the existing legal, scientific and institutional infrastructure. Often optimal solutions from an ecological perspective will encounter feasibility, applicability and cost constraints and the specific combination of management and regulatory tools needs to be tailored to the specific context of each country, region or locality.

Table 5.3 Assessment of management and regulatory tools

Robustness	The robustness of offsets will depend crucially on the arrangements for long term management set out in the offset management plan, the adherence to these arrangements by the offset provider, and the ability of the regulatory authorities to enforce these requirements. This in turn will depend on effective monitoring and regulatory oversight. Accreditation and certification systems can enhance the robustness of offset delivery by ensuring that providers meet appropriate standards, and by incentivising the long term adherence to these standards. The robustness of regulatory systems will be enhanced by integrating offsets into existing systems of planning and environmental regulation, where these

	have the capacity to ensure that offsets meet their required objectives.
Durability	The durability of offsets will depend on long term adherence to the management plan, supported by ongoing monitoring and regulatory scrutiny, and backed by the ongoing threat of enforcement action in case of non-compliance.
Flexibility	The flexibility of offset systems will depend to some degree on existing legal and institutional infrastructure in place for other conservation activities Integrating offsetting within regional regulation and management can add greater flexibility to implementation options for developers - allowing offsetting to be properly integrated alongside other options in the mitigation hierarchy. Flexible local systems may also allow offsets to be planned and delivered in a strategic way that maximises local benefits and the development of ecological networks. Contractual arrangements for offsets, and the management plans that support them, can be flexible to local conditions in order maximise conservation benefits.
Applicability	In an EU context, applicability of regulatory and management tools will depend on the (administrative) level and (ecological) scope of offsetting. In some countries, integrated regulatory and management capacities at the regional or municipal level will support development of offsets aligned with local conditions. In others, with more centralised systems of planning and conservation management, more standardised and prescriptive rules may be appropriate.
Feasibility	Feasibility is a major concern with regard to management and regulatory tools in a number of countries. Access to ecological data, land and financing are prerequisites for effective offsets. In most cases, offsetting systems have built on existing resources developed over time, although a national regulation has been critical in creating demand. In many countries, extending existing regulation to enable offsetting will prove significantly more feasible than new legislative measures. Benefits of extending existing regulations include greater regulatory capacity and familiarity with compensation measures, as well as established responsibilities for monitoring and enforcement.
Cost	The cost of providing offsets is highly sensitive to economies of scale, and regulation has been shown to be pivotal in supporting demand in this regard. Upfront capital requirements are generally significant, although small in relation to the total cost of large infrastructures projects. Effective measures to secure long term benefits, including management plans, performance standards, accreditation and certification arrangements, and long term monitoring and enforcement, all entail significant costs and require sufficient resources to be invested in offset design, implementation and management. Effective regulation and enforcement of offsets, will entail additional costs on public authorities. It is important that these costs are recouped from developers to ensure observance of the polluter-pays-principle. Integrating offsets into existing regulatory systems – wherever effective and appropriate – will often be more cost-effective than establishing new regulatory systems and bodies.

5.2.2 Securing land use

For a functioning offset system to deliver durable long term benefits it is essential that rights can be secured for offset sites that enable their use to be guaranteed over the long term, that appropriate management can be implemented that enables the conservation goals to be met, and that an appropriate institutional infrastructure is in place to administer this.

There are a number of options for delivering on each of these land-related requirements, although they are not necessarily mutually exclusive and can be combined in different ways. The principal options are identified and assessed below including

- Land acquisition and leasing;
- Management agreements;
- Conservation covenants/easements;
- Offset registries; and
- State and NGO stewardship.

Table 5.4 Assessment of mechanisms for securing land use

Robustness	A system that uses a combination of the identified mechanisms (to secure access to land and to secure its future for conservation purposes) provides the highest level of robustness, enabling securing of title, land use type and detailed management as well as a transparent and accountable institutional arrangements. Acquisition provides the strongest guarantee of securing rights to land. Where land is not acquired or leased, conservation covenants/easements and management agreements will be entered into voluntarily and require an appropriate level of compensation to be paid to secure to the land owner. A comprehensive offset registry can help to avoid the multiple use of land for offset projects or unintended detrimental impacts from unrelated future projects. Making offset registries publicly accessible can allow for public scrutiny of outcomes (and activities) by civil society groups, which can aid successful offset delivery over the long term. Environmental NGOs are potentially good stewards as, for the appropriate NGOs, the conservation or remediation of land aligns with their stated objectives. However in MS where the NGO sector is small or lacks capacity it is likely to be harder to identify NGOs that could adequately assume the role. NGOs are not accountable to citizens and therefore some form of State oversight is necessary to ensure adequate performance. Where the state has appropriate capacities and has appropriate legislative commitments to biodiversity offsetting, it should be considered to be a robust steward.
Durability	Acquisition provides for rights in perpetuity (but does not itself prevent the future sale of the land). Acquisition can be paired with conservation covenants/easements. These are legal instruments attached to the land rather than an individual. Thereby they 'run with the land' providing greater security over the long term i.e. in perpetuity. In the case of a sale of the site or bankruptcy of an offset provider the offset obligations remain pertinent and secure. Long term management agreements (over decades with ongoing duties of maintenance, although not necessarily in perpetuity) may be able to be negotiated for offsetting but are often made with individuals (rather than areas of land) and therefore may be terminated in the case of the sale of land. Utilising acquisition and conservation easements/covenants can overcome some of the issues associated with relying solely on management agreements. Entry into a registry provides a mechanism for tracking obligations and offset

	delivery over time, ensuring that duties and performance are recorded and visible. Legal duties associated with land titles (e.g. linked to leases or covenants) are typically recorded in national land registries. NGOs may be less subject to political lobbying and policy fluctuation than the State. However they will have their own political and financial pressures. Safeguards need to be put in place in order to determine what happens in the event that an NGO ceases to exist or goes bankrupt.
Flexibility	Acquisition can be undertaken at a variety of scales and locations. A key constraint however will be whether there is a sufficient supply of land which can be purchased. It should be noted that supply of available land may be constrained (and EU evidence suggests that it usually is) by a number of factors including: price (and available acquisition budget), permanence of existing uses, suitability of site ecology for offset-related remediation, as well as willingness of land owners to sell. Ultimately therefore the ability to purchase land varies across localities within countries. Leasing can be used as way of circumventing some of these barriers to directly securing rights. Voluntary entry of land under a conservation covenant/easement and/or management agreement can also be used to overcome a number of land market supply barriers and secure land use and management without the need for acquisition. The detail included in a conservation covenant/easement and management agreement can be tailored to the particular situation, providing flexibility across sectors and locations (although there is a trade-off between flexibility and transaction cost). Independent offset registries can be designed to allow for the desired level of flexibility. For stewardship, flexibility will be in part a function of the capacity and availability of appropriately aligned institutions.
Applicability	All EU MS have functioning land markets and EU laws provide some basic harmonisation across them. Whilst covenants and easements are currently utilised in land markets across the EU, new conservation covenants/easements would need to be established that overcome certain restrictions that are currently in place and make covenants/easements appropriate for use in an offset system. All EU MS could set up functioning offset registries. All EU MS have existing land registries (and/or cadastres) into which law typically requires that all new ownership be registered, though this is not likely be sufficient for offset tracking and may not be particularly visible or transparent.
Feasibility	There are no fundamental technical or administrative barriers to securing rights through normal market mechanisms and most EU MS have experience of negotiating management agreements (e.g. for agri-environment schemes). Establishing new conservation covenants/easements in EU MS will require appropriate administrative arrangements, legal arrangements and production of guidance. Appropriate administrative capacity is required for negotiation of all agreements, for communication of management requirements and for developing and maintaining an offset registry over time. The adequacy of NGO capacity varies between EU MS and across NGOs within individual MS. In some MS there is a strong track history of NGOs managing land. It should also be recognised that historical norms and public preferences play an important role in determining feasibility, with differing MS having pre-conditioned preferences and tolerances for

	administration/involvement by NGOs and by the State
Cost	Land acquisition is likely to be the most expensive (if the most secure) option for securing land rights. For this reason, many MS which had employed land acquisition in relation to Natura 2000 sites have reduced or stopped this practice, although affordability and cost issues may be framed differently depending on whether the burden falls on the State or a private developer or offset provider. As land purchase is typically an upfront cost it requires appropriate access to finance, which has a real financial cost. Where land is not acquired or leased, conservation covenants/easements and management agreements will be entered into voluntarily. This can be of particular importance where land ownership is concentrated in the private sector and securing access via other means (i.e. acquisition or leasing) would entail potentially large up front capital costs. However voluntary entry into conservation covenants/easements and management agreements requires compensation to be paid the land owner for profit foregone (i.e. of using the land for another use). If land prices incorporate considerations of future profits then one might assume that the discounted costs of land acquisition would be similar to the discounted costs of ongoing compensation payments. There will be a variety of costs involved in establishing the legal framework for conservation covenants/easements in EU MS, including for example legal negotiations and production of guidance. There will also be ongoing transaction costs associated with the use of conservation covenants/easements and management agreements. Establishing standardised covenant/easement texts combined with separately detailed management plans would help to limit transaction costs. Negotiation of multiple agreements, design of multiple bespoke agreements and renewal of agreements will increase the administrative and transaction costs. The development and ongoing maintenance of an offset registry will place a cost burden on the responsible organisation.

5.2.3 Sustainable finance

Securing access to finance for offsetting is necessarily a long-term proposition, requiring a reliable and long-term payment vehicle to support capital investment and ongoing management activities. Internationally, trust funds have emerged as a preferred design option for ensuring good governance and long term fiscal management of offsets, and a mechanism to disburse funds to pay for offset activities over the years.

Securing the long term future of offsets depends on safeguarding against the risk of potential future failure. This may be achieved through appropriate guarantees or securities. Insurance pools have become increasingly popular to safeguard mitigation projects in the USA against the risk of failure. These pools cater to the specialist nature of offsets (thus commanding lower premiums) and the need for protection against legal challenges and other foreseeable costs. Specialist environmental insurance pools have emerged in a handful of countries including France, Italy and Spain, similar to the insurance pools administered for offsetting in the USA.

Risk multipliers (requiring increases in the extent of offsets required to provide a margin for failure – see section 3 on metrics above) provide an alternative form of insurance.

Table 5.5 Assessment of mechanisms for sustainable finance

Robustness	Trust funds can offer secure long-term conservation benefits by underwriting the cost of ongoing management activities. The most robust models will finance management activities through interest payments - otherwise the fund is ultimately a declining resource over the long term. Including contingencies

	will also enhance robustness. Insurance pools offer substantial additional security for conservation benefits over the long term, ensuring that the liable party in the case of offset failure has an identified source of revenue to fund management costs in the case of failure of the offset provider. Insurance pools have the added benefit of providing security to offsets of different sizes. Risk multipliers may also provide an effective form of insurance against future risk, providing the level of risk is understood. However, underestimating risk could lead to a failure to deliver no net loss.
Durability	The durability of trust funds is contingent on a number of factors, including the specific structure, the level of investment and holder of the fund. Experience from the US points to conservation groups becoming involved in funding offsets without fully understanding their financial liabilities. Insurance pools can enhance the durability for offsets, by spreading the risk of investment and lowering risk premiums compared to individual insurance arrangements, as well as providing protection against other costs such as legal challenges.
Flexibility	Trust funds take a number of forms depending on the distinct context of each country, its financial institutions and the scale of financial provision required. Inclusion of a contingency when specifying the size of the fund will enhance flexibility to changing circumstances. The scale and diversity of environmental liability insurance available across the EU would seem to indicate that the EU insurance industry is well-placed to deliver the range of specialist insurance products that provide security for mitigation projects in the USA. Risk multipliers can also be applied in a flexible way, to account for variations in project risk.
Applicability	There are no fundamental legal issues regarding the establishment of conservation trust funds in EU MS, although the legal system is likely to influence the legal structure of a fund. Trust funds, foundations and associations could be adapted for use across the EU. At present, environmental insurance in various forms is available in most EU countries. Amounts vary according to domestic demand, but cater to the distinct needs of SMEs and large corporations in different countries. Cover is generally available for all environmental risks, including primary remediation, compensatory remediation and complementary remediation.
Feasibility	Access to environmental finance remains a major challenge for many countries, and the relatively high risk profile of offsets presents a further barrier to uptake. Nonetheless, development of relevant regulation could be expected to support a growth in supply of trust funds for offsetting in the EU. Experience from the USA, together with the emergence of ELD insurance products, appears to indicate that insurance products for offsets are likely to develop rapidly where regulation is in place to support demand. Unlike offsets, insurance products can potentially be traded between Member States, helping to address potential supply constraints in some MS.
Cost	Setting up trust funds for offsetting typically requires access to significant amounts of capital (particularly in cases where the fund is expected to cover the entire lifetime management costs). Nonetheless, overall capital requirements are modest in comparison to many large infrastructure projects, and conservation banks can offer economies of scale for small-scale impacts.

	Overall insurance against all manner of risks is likely to significantly increase the durability of offsets but increase costs. If a risk is already insured for through metrics, regulators should examine whether it is reasonable to require additional insurance.

6 Conclusions and Policy Implications

6.1 Offset metrics

These conclusions draw on the preceding analysis of metrics and their ability to contribute to the EU no net loss initiative and related environmental objectives, including the full implementation of the Birds and Habitats Directives. The principal conclusion from the analysis, as outlined in section 3.1.3, is that on the basis of the advantages and disadvantages of the various options for metrics (including associated multipliers and exchange rules), **the best approach, in the short-term, would be to develop common principles on the development and use of metrics, perhaps accompanied by suggested important features of different metrics that could suit different situations**.

The following section provides some recommendations on some key principles that could be further considered.

1) A set of metrics should be used that reflect the differing levels of importance of the various biodiversity and ecosystem services affected by developments and other activities, and the risks that offsetting residual impacts on them may result in uncompensated biodiversity losses. Thus the data and analytical requirements for offsets should be proportionate to such risks.

Accordingly, losses and gains of habitats of high importance (including habitats of Community interest as listed in Annex 1 of the Habitats Directive) should be measured with metrics that include detailed and reliable consideration of habitat condition. Thus, because such habitats normally require like for like offsets, a detailed area x condition only metric would be suitable for measuring such losses and gains. For example, the habitat hectares metric as developed in Victoria, Australia (see Annex 1.1) would be appropriate provided that the basis of condition assessment is based on sound ecological principles and the best available empirical evidence, and thorough field assessments of condition are carried out by adequately trained ecologists. Condition assessments should include consideration of the spatial characteristics of the impacted site and proposed offset area and their ecological implications (e.g. relating to connectivity).

It would be appropriate for metrics covering Annex 1 habitats to use consistent condition criteria for each by biogeographical region though taking into account more local variations. A similar approach should be taken for habitats that are of high national importance, such as those listed in national biodiversity action plans.

It would be appropriate to use a simpler metric for measuring losses and gains in habitats of moderate potential value. However, because trading up may be appropriate for such habitats, metrics are required that also consider the potential biodiversity value of the habitat, so that different habitats can be compared. Thus some form of area x value x condition metric is appropriate for such habitats. This could be similar to that developed by Defra for use in the pilot offset trails in England (see Annex 1.2) but should have more than three distinctiveness (i.e. value) and condition classes (for which the current English metric has been criticised). It is also recommended that standardised biodiversity values are based on empirical evidence and adequate consultation amongst experts and stakeholders. Care should be taken to ensure that the range of value weightings is appropriate to the range of habitat values, thus ensuring that the importance of high value habitats is properly reflected. The habitat typologies and ascribed values for each habitat type should also be applicable to as large an area as appropriate in ecological terms, to avoid the development of numerous currencies with arbitrary variations, because this can lead to problems where development projects cross administrative boundaries (as seen in Germany). The assessment of condition should also always be based on adequate methods that are fit for their purpose.

For low value habitats (which would not include any semi-natural habitats) it may be appropriate to use simpler metrics that are based on standardised general or potential values, with adjustment factors where necessary to take into account local variations or spatial factors (such as those widely used in Germany). Values should be agreed as for value and condition metrics described above.

Where necessary, the metrics listed above should be supplemented through the use of appropriate species focused metrics. For example, the Habitat Evaluation Procedure (HEP) type metric developed in Somerset, England (see Annex 1.2) could be suitable for wider application in the EU. Although it requires a good understanding of the ecology and distribution of species of conservation importance, the collation such data in advance of developments can be cost-effective because areas of high biodiversity importance can be avoided at early stages of the planning process.

Similarly habitat metrics that focus on biodiversity values should be supplemented by the use of appropriate ecosystem service metrics, such as those used in Germany and the United States. Because ecosystem services are highly context specific and therefore their importance varies, it is unrealistic to attempt to develop standardised ecosystem service metrics. Furthermore the development of ecosystem service metrics is still at an early stage and further work, for example relating to the MAES initiative, is required before recommendations on best practice ecosystem service measurement can be made.

2) Multipliers should be used where necessary to adjust metrics according to potential risks of offset underperformance (and other uncertainties) and the need to compensate for time delays in the provision of biodiversity gains from offsets.

Risk multipliers should be based on good empirical evidence and take a precautionary approach. However, additional steps should also be taken to avoid failure in high risk offsets, such as through hedge betting strategies and contingency measures. In other words, risk multipliers should not be the sole means of managing risks.

The use of averted risk offsets (i.e. protection and management of threatened habitats) in Europe needs to be carefully considered because of the difficulty of ensuring their additionality, as well as other drawbacks (Tucker *et al*, 2014). However, if they are to be used then outcome modifiers (or endgame modifiers) that are based on an assessment of the conservation status of the targeted habitats or species should be used to ensure their adequate protection in the long term (i.e. when all areas of habitat are either protected or developed).

Time multipliers should be used in cases where there is a delay between the losses incurred and compensating gains. These can be calculated using an appropriate discount rate (as in England). This can significantly increase offset requirements if there is a lengthy time gap between losses and gains.

3) It is essential that metrics are used in conjunction with clear exchange rules that take a precautionary approach to ensuring no net loss (or agreed net gain objectives). In this respect exchange rules should ensure that offsets are based on 'like for like' or 'better' rules both in terms of general or potential habitat value and condition. Exchange rules are also required to ensure the appropriate location of offsets, which should normally be close to impacted site, but other factors such as spatial connectivity and viability of the offsets site should be taken into account. They can play an important role in ensuring that ecosystem services are not lost.

Trading up should be encouraged where this can contribute to agreed strategic conservation objectives such as habitat restoration, green infrastructure, ecological network and protected area network goals. Similarly, where appropriate, offsets should be placed in strategically beneficial locations identified through proactive ecological surveys and planning in consultation with stakeholders.

4) The development and use of offset metrics needs to be underpinned by an appropriate policy and legislative framework, and adequate institutional support, as for example set out in the recent no net loss policy options study (Tucker *et al*, 2014).

Adequate institutional support is required in particular regarding the ability and capacity of environmental authorities to assess offset proposals and the appropriate use of metrics. At the impact assessment and offset planning stage, adequate resources are required to enable authorities (or contracted accredited certifiers) to carry out field checks of offsets and metric measurements (e.g. through a risk-based sample approach) and where necessary take actions to enforce recalculations or adjustments. Authorities should also check that the metrics have correctly measured the actual biodiversity and ecosystem service losses and gains resulting from the development and the offset, such that additional offset measures are taken if it is necessary to achieve no net loss or other the agreed objectives (e.g. in terms of net gains). Such checks should be carried out at appropriate intervals over the long-term.

Institutional support and guidance should also be provided by authorities to developers and consultants, in order to ensure metrics are adequately understood and properly used. Furthermore impacts on biodiversity and ecosystem services from development projects can be avoided and related project costs and delays minimised through the provision of advice by ecologists and other specialists in national and local authorities. The costs of the provision of such advice is likely to be very low compared to the overall costs of a project and therefore investment in the capacity of authorities to support well-planned offsetting and the strategic collation of data on biodiversity and ecosystem services can be cost-effective in overall economic terms.

6.2 Mechanisms for securing long term conservation benefits

This section offers conclusions on the desired characteristics of an offset regime that contribute to securing long term conservation benefits, and the likely challenges in ensuring the implementation of appropriate mechanisms across the EU under the broader banner of the EU NNL initiative.

6.2.1 Desired characteristics of an offset regime

Ensuring that benefits are secured over the long term requires a suite of appropriate mechanisms to be put in place which ensure that the chosen activities are delivered in an effective, sustained and measured way. In particular there is a need for:

- A robust regulatory requirement and management regime;
- Security of land use; and
- Sustainable finance.

The mechanisms used to address these three elements should satisfy a series of basic characteristics. Table 6.1 provides a consideration of these possible mechanisms and their characteristics.

Table 6.1 Mechanism characteristics

Long term mechanism types	Characteristics
Mechanisms for regulation and management: A legal framework and biodiversity offset management plans	■ Provide for a binding, legally enforceable contract ■ Suited to the institutional capacity of the MS ■ Ensure for effective standards, compliance and accountability ■ Balance ecological rigour with technical feasibility ■ Link to long-term socioeconomic and land-use planning objectives
Mechanisms for securing land use and management:	■ Enable both security of access to land for offsetting purposes *and* appropriate

Long term mechanism types	Characteristics
Contracts and management agreements for the delivery of offset activities by third parties coupled with covenants/easements or landuse restrictions to ensure management is required in perpetuity and 'runs with the land' where there is a change of title	management of that land ■ Provide for security over the long term, ideally in perpetuity ■ Provide for flexibility in year-to-year land management whilst limiting ongoing transaction costs ■ Utilise appropriate custodians and ensure transparency and opportunity for external scrutiny as well as appropriate safeguards for underperformance or organisational failure
Mechanisms for financial sustainability: Trust funds or other financial tools that enable the good governance and management of offset investments. Insurance products or risk management mechanisms.	■ Provide for financing of upfront costs ■ Provide for ongoing financing of offset management ■ Include contingencies for unforeseen costs ■ Provide financial insurance for offset failure (where the risk of failure are not built into the metric design) ■ Supported by appropriate institutional arrangements for administering funds

6.2.2 Implementing offset mechanisms

Following the research on, and assessments of, mechanism options undertaken for this study, it is useful to consider the implications for biodiversity offset policy implementation at the EU level.

Regulatory and management systems

Outside of the Natura 2000 network, only a few Member States (notably Germany, France, the Netherlands, Sweden and the UK) have developed or are developing regulatory systems for biodiversity offsetting. This is consistent with findings from ICF's previous study on Habitat Banking in the EU, which suggested that compensation beyond regulatory requirements (e.g. the Natura 2000 network) is rare owing to technical feasibility and cost considerations, the contentious nature of such compensation, as well as a general lack of regulatory drivers to support demand.

The international and EU examples outlined in this research attest to the range of institutional and management arrangements possible for delivering offsets, but in general, the most successful examples have benefited from early engagement and consultation with affected stakeholders. Whilst regulatory systems need to enforce ecological rigour, they also need to allow sufficient flexibility to ensure that offsets are viable 'on the ground'.

Whilst regulatory systems need to enforce ecological rigour, they also need to allow sufficient flexibility to ensure that offsets are viable 'on the ground'. To achieve the correct balance between ecological rigour and flexibility of application is difficult even at the level of individual Member States: establishing detailed rules that can be applied consistently but flexibly across the 28 Member States of the European Union would be extremely challenging. However, a policy framework, setting out key principles for ensuring long term sustainability of conservation benefits could offer a more pragmatic way forward.

Access to land

Access to land and securing management obligations can be achieved through market and accompanying legal mechanisms. The most appropriate approach to ensuring that adequate access and obligations are secured over the long term will vary across MS and within different regions and situation within MS. Suitable flexibility will need to be incorporated into any policy to allow national and local contexts to be taken account, whilst ensuring that the mechanisms employed will provide minimum level of robustness.

Land acquisition provides the greatest level of security of access, however is likely to be the most problematic in terms of land availability and financial cost burden. Because of these problems a general trend away from public land acquisition has been witnessed across MS in relation to Natura 2000 sites. However the cost burden of offsets will fall in many instances on the private sector as it is the developer requiring the offset who pays. As such land acquisition is likely to be an important tool for securing access, most notably where securing access by other means is not possible and where the strength of security offered is considered necessary. This may be the case where there are significant competing pressures for land, or where security of title is a pre-requisite for other mechanism e.g. securing finance. Despite EU rules that apply to MS land markets, there can be constraints imposed by MS that limit how land can be used or how easy it is to change its use. Most notably this is the case in the agriculture sector. Further issues surround land valuation and ensuring equitable markets.

Management agreements have been successfully employed to secure management obligations for Natura 2000 sites and have aided a shift away from expensive land acquisition. However it is unclear whether they can be so widely used to underpin an offset market without any form of additional legal safeguard. There is evidence that the use of conservation covenants or conservation easements can provide that legal safeguard, securing land use obligations, which may or may not also require land acquisition or leasing, providing the potential for security in perpetuity. Traditional covenants and easements are often not satisfactory for this purpose and explicit conservation covenants or easements would need to be established in MS, along with an appropriate institutional infrastructure to determine an appropriate benefit holder and ensure sufficient oversight and safeguards. Allied with these, management agreements can then provide a more flexible tool for stipulating detailed management requirements.

Sustainable finance

Financial mechanisms to support the long-term delivery of conservation benefits from offsets may take a number of forms, and in the context of biodiversity offsets can be divided thematically into ‘mechanisms to ensure sufficient capital’ and ‘mechanisms to safeguard against risks of failure’.

Conservation trust funds provide an internationally accepted means of financing offsets in the long term, and, other than the costs of financing them, there do not appear to be significant barriers to their application in the EU.

Safeguards against risk can be secured through financial insurance, and/ or through the application of risk multipliers that increase offset requirements (and therefore allow a margin for failure). There is a need to examine insurance against risk across the system as a whole, and to avoid “over-insurance” which could entail excessive costs.

Experience from the extractive sector (Mining Waste Directive) and industrial waste management (Environmental Liability Directive) points to the importance of ensuring enforcement of remediation and compensation liabilities and requiring funds to safeguard against management costs in the case of project failure. In each case, securitisation ensures that the public sector is not exposed to unlimited liability from the failure of offset provision.

Europe benefits from a large and liquid environmental liability insurance market but there is a growing trend towards mandatory insurance requirements in law. This increases direct costs to industry but may help lower premiums as demand increases. The complex nature of offsets - which include ongoing liabilities – may initially entail higher risk premiums than other forms of environmental liability insurance but this may be offset by growing demand under new regulation and resulting economies of scale. Insurance pools are one example of how to accelerate this process and have been successfully applied in US mitigation banking.

Such pools already exist in Spain, France and Italy, and have been highlighted by the UK government as having particular potential for supporting the development of offsets. Studies by the insurance industry point to the key role of regulation in supporting the growth of environmental liability products, but it is possible that additional incentives may be needed to ensure that these products ensure the maintenance of long-term conservation benefits. In the USA, for example, security requirements are informed by the track record of the provider in offset delivery, and in some cases, are reviewed periodically.

In the EU, the Commission is currently exploring a mandatory requirement for liability insurance in connection with the ELD. Whilst this is expected to further increase demand for such insurance products, the proposal is only viable because of the relative maturity of the ELD insurance products market and competitive premiums. As a new and complex prospect, requiring liability insurance for offsets may add substantially to the overall costs of offsets. Endorsement of insurance pools may be a more desirable alternative to promote the early development of offset markets.

6.3 Evidence gaps and priorities for further research

While a number of studies and evaluations address the theoretical and basic design requirements of biodiversity offsets, there are comparatively few which provide detailed consideration of the mechanisms required to secure benefits over the long term. This is an important gap because the evidence reviewed in the course of this study and through consultation with stakeholders points to uneven implementation and significant discrepancies in quality management for common regulatory systems. Clearly, a balance must be found between systems that are suitably prescriptive to establish common minimum standards for maintenance of long-term benefits, and systems that are realistic and achievable, as well as those that can be maintained over time.

In general further testing of the applicability of different mechanisms in MS would help to provide an understanding of the potential gap of existing infrastructure and capacity to deliver a fully functioning offset system. This would help to inform how the policy is implemented by the EC as well as aid individual MS understand their state of readiness. In particular there is a need for real world evidence accumulated through pilot projects. In France and the UK, pilot projects have come up against significant barriers that have inhibited their performance. Lessons can be drawn from these and new pilot studies. This is necessary to help understand the intricacies of how a system should be designed, but crucially to also begin to develop a ‘proof of concept’ that can provide the foundation for broader development of a broader offset market.

Securing measurable benefits over the long term is a pre-requisite for an offset system to contribute positively to NNL. A rapid expansion of the offset market will require relatively low cost and legally robust mechanisms to be used to ensure land obligations are delivered over the long term. In the US conservation easements are an established mechanism for securing offset obligations over the long term, coupled with management agreements and backed by instruments such as insurance. In the EU, the UK and France are experimenting with the use of conservation covenants, which will require new legal arrangements. The development and performance of these new conservation covenants should be evaluated to understand how well they perform, where further legal definition and supporting technical and institutional arrangements are required, and how transferable the principles and detail are to other MS.

Financial mechanisms have been identified as a key design element of offsets and a wide range of stakeholders have been consulted in the course of this research. However, even in well-established systems (such as US Wetland and Conservation Banking) there is a significant degree of uncertainty and a lack of comprehensive analysis around financial management of offsets. Many experts and practitioners were unable to provide general commentary on financing of offsets because of the range and complexity of arrangements (and because some of the arrangements are commercially confidential). This is a crucial gap, because many authorities tasked with evaluating offset design and implementation may lack the specific expertise or financial acumen necessary to appraise the financial management of offsets. In the EU, access to capital is likely to be a major constraint on offsetting because of a lack of investor awareness and familiarity with offsets in the private sector, and an overall lack of financial resources amongst many public authorities to support offsetting. A specific gap analysis, analysing the specific capacities of public, private and non-governmental organisations to secure financing for offsets, as well as opportunities for innovative financing models, would have some utility in this regard. For ongoing management and feasibility aspects of offsets, it would be important to understand the capacities of existing regulatory bodies to address long-term financial planning aspects (e.g. discount rates, non-declining endowment trusts) alongside ecological management criteria.

To achieve no net loss, offsets are required to deliver measurable conservation benefits in perpetuity. However, even in the US and Australia, where offsetting experience is most established, offsets are relatively new and best practice is still emerging. We do not therefore have the experience to know how durable offsets are over the long term, and how well each of the mechanisms performs against its stated aims over the long term. There are examples in early US schemes where insufficient safeguards have resulted in failure to achieve no net loss. This has informed improvements in practice and the development of more stringent safeguards. However, the long term effectiveness of mechanisms available to secure long term conservation benefits cannot yet be fully evaluated. Ongoing monitoring and evaluation of these mechanisms is needed to assess the long term benefits of offsets, and hence their ability to achieve no net loss of biodiversity and ecosystem services.

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Annex 1 Use of Metrics in Australia, England, France, Germany, South Africa and USA

A1.1 Use of metrics in Victoria, Australia

A1.1.1 Regulatory framework and offsetting requirements

Australia has a number of regional offsetting schemes (Madsen *et al*, 2010) of which the scheme in Victoria of particular interest as it pioneered the development of the habitat hectare metric. The regulatory framework for the scheme in Victoria is therefore outlined here.

Until 2013, the main driver for offsetting in Victoria was the 2002 Native Vegetation Management policy, which aims to achieve no net loss in accordance with the mitigation hierarchy. Under the Victoria Planning Provisions native vegetation is comprised by plants that are indigenous to Victoria (including trees, shrubs, herbs and grasses) and a planning permit is required to remove, destroy or lop native vegetation unless the removal is the result of a use that is not regulated by the planning scheme[28]. If a permit to remove native vegetation is granted, an offset which makes an equivalent contribution to Victoria's biodiversity will be required. Offset requirements were determined in accordance with *Permitted clearing of native vegetation – Biodiversity assessment guidelines*.

The guidelines detailed impacts that must be offset (and which impacts must or should be avoided), 'like-for-like' conditions, and requirements for the proximity of offset relative to the impact site. Neither impacts nor offsets are allowed in areas of 'very high' conservation significance except in 'exceptional circumstances.' Clearing in 'high' or 'medium' areas of conservation significance is generally not permitted, but some clearing may be permitted in areas of 'low' conservation significance.

Offsets can be one, or a combination of the following:

- improvement in the condition of existing vegetation with ongoing management and protection using an appropriate security arrangement;
- re-vegetation of a site with ongoing management and protection using an appropriate security arrangement, although there are restrictions on when re-vegetation can be considered an offset.

Offset requirements may be defined in terms of three currencies: vegetation or habitat (ie a condition x area metric; 'large old trees' and 'new recruits' (i.e., tree planting). The first of these metrics is based on area and site-quality measured by the 'habitat hectares' methodology (see below). These credit types are based on Ecological Vegetation Classes (EVCs) within Victoria's 28 bioregions, accounting for 2,500 possible types of EVC credits. Developers can deliver offsets on their own land or purchase a native vegetation credit from a third party, provided that they meet standards set out in *The Native vegetation gain scoring manual*[29].

In September 2012, the Victorian Government announced a review of Victoria's native vegetation permitted clearing regulations, the aim of which was to improve and strengthen the regulatory system to deliver better outcomes for the environment and the community[30]. As a result of the review, the native vegetation permitted clearing regulations have been reformed, in order to:

- provide a stronger focus on the value of native vegetation for state-wide biodiversity;

[28] Department for the Environment and Primary Industries http://www.depi.vic.gov.au/environment-and-wildlife/biodiversity/native-vegetation/native-vegetation-permitted-clearing-regulations/requirements-for-a-permit-to-remove-native-vegetation

[29] http://www.depi.vic.gov.au/__data/assets/pdf_file/0005/198968/Gain_manual_NVR.pdf

[30] http://www.depi.vic.gov.au/__data/assets/pdf_file/0007/180637/Overview_NVR.pdf

- reduced regulatory burden for landholders while at the same time providing upfront information about the value of native vegetation on their land; and
- improve decision making.

The reformed system is now being implemented, following changes to the Victoria Planning Provisions in September 2013, with the four key reforms aiming to:

- clarify and amend the objective of the permitted clearing regulations;
- improve how the biodiversity value of native vegetation is defined and measured;
- improve decision making; and
- ensure offsets provide appropriate compensation for the environment.

A1.1.2 Habitat hectares metric, Victoria Australia

The habitat hectares metric initially developed by Parkes et al (2003) and updated by the Victorian Government Department of Sustainability and Environment (DE, 2004), provides a way of calculating losses and gains in vegetation condition for each distinct EVC, based on units of measurement that take into account the area affected and the quality or condition of the vegetation impacted. These are described in a 'benchmark' that sets out at least 10 types of habitat attribute for each EVC, such as: number of large trees, canopy cover, number of understorey lifeforms, cover of weeds, recruitment, cover of organic litter, abundance of logs, patch size, proximity of remnant vegetation and distance to core area. The attributes in the benchmark are weighted according to their significance to the overall condition of the system. A user measures each attribute at the impact site before the impact and the predicted score after the impact, comparing the measurements against the benchmark which represents the pristine condition of the habitat in question. The scores for each attribute are then added (according to their weightings) to provide an estimate of the site's condition expressed as a percentage pristine condition. The area of the habitat is then multiplied by this percentage change in condition. The same approach is used to estimate the gains at the potential impact sites, comparing the actual measurements before the offset activities start with predicted realistic outcomes from the offset, again compared with the benchmark levels.

Put most simply, the loss of 100ha of forest at '50% quality' is expressed as the metric of 50 "habitat hectares" and can be compensated for with offset gains of 50 habitat hectares. This can be achieved, for example, through the gain of 25% of 'condition' (=quality) over an area of 200ha, or 100% 'condition' over an area of 50ha. Restrictions can be set so that gains in low condition areas cannot be substituted for losses of high condition areas, and also so that gains in one attribute (eg numbers of a particular species of tree) cannot be substituted for another attribute (eg some of the landscape features like patch size). The metric was originally designed to focus on habitat structure, and thus provide proxies for composition and function. In practice, some aspects of composition and function have been included as attributes and are thus measured directly. The attributes can be chosen to represent particular species of value, if necessary.

The Victorian Habitat Hectare system is a relatively sophisticated system that is based on some fundamental ecological principles. The rationale for including habitat condition in offset metrics is widely acknowledged but estimating ecological condition objectively and reliably is not easy. Consequently, the habitat hectares system has been criticised, notably by McCarthy *et al* (2004). A key problem is that the quality attributes are compared with benchmarks that are long undisturbed areas that attempt to represent stable climax vegetation communities. However, McCarthy *et al* point out that current ecological thinking is that many vegetation communities continue to change and that good quality habitat is often linked to recurrent disturbances. Thus they suggest that disturbance regimes should be considered in the assessments.

McCarthy *et al*. also point out that the assessments of the attributes can vary greatly between assessors and therefore this needs to be taken into account. Also they question the rationale for simply multiplying the habitat area by the quality score, especially as condition will be affected by the habitat area (e.g. with respect to the species-area relationship). Another fundamental problem is that, like many metrics, attribute values are added which implies that the attributes are substitutable. For example, important detrimental changes such as the loss of trees can be masked by less important improvements such as increased deadwood. McCarthy *et al*. suggest that this problem can be solved by using a weighted multiplication approach to reflect the varying importance of the attributes. Gardner *et al*. (2013) note that the problem can also be solved through use of disaggregated currencies or, at least in part, by establishing exchange rules that set minimum values (and possibly upper limits) to which the individual components that make up an aggregated currency can be substituted.

The procedures for the permitting the clearance of native vegetation have been revised and new Biodiversity Assessment Guidelines[31] produced by the State of Victoria Department of Environment and Primary Industries (DEPI) in 2103, which have been incorporated in all planning schemes in Victoria. The new procedures retain the habitat hectares metric, but this is in the process of being revised[32]. As before the condition of native vegetation is measured at a site but condition scores are also modelled across the landscape based on survey data and environmental variables. These scores have been mapped across Victoria and can be found in the native vegetation condition map included in the Native Vegetation Information Management system, found on the DEPI website.

However, the habitat hectares metric is now multiplied by a strategic biodiversity score (which quantifies the site's importance for Victoria's biodiversity relative to other locations across the landscape) to produce a biodiversity equivalence score. The strategic biodiversity score is derived using a spatial prioritisation tool that ranks locations in Victoria for their conservation priority on the basis of rarity and level of depletion of the types of vegetation, species habitats, and condition and connectivity of native vegetation. The scores can be obtained for all vegetation from a Strategic Biodiversity Map found on the DEPI website.

However, if a site is important for a rare or threated species, then the habitat hectares metric is multiplied by a habitat importance score (which is a measure of the importance of the location in the landscape for the persistence of the particular rare or threatened species).

Offset requirements and procedures also vary according to their risk to biodiversity which is assessed according to 3 levels of extent of risk (i.e. scale of impact) and location risk (ie whether it occurs in areas of high low, medium or high biodiversity importance).

A1.1.3 Conclusions

Despite some limitations, the habitat hectares approach remains a more rigorous approach than area x value metrics and is reasonably quick and pragmatic form of assessment to apply that gives accounts for changes in a broad range of biodiversity components. It continues to be used in Victoria[33] (although it has been revised and is currently being thoroughly reviewed) and has been widely adopted and adapted internationally, for example in Western Australia (Hajkowicz *et al.*, 2009), South Africa (Kotze, 2005) and at a suite of projects worldwide associated using the BBOP guidance. Some adaptations have attempted to deal with some of the problems outlined above, whilst others have incorporated other factors into the system.

[31] http://www.depi.vic.gov.au/__data/assets/pdf_file/0011/198758/Permitted-clearing-of-native-vegation-Biodiversity-assessment-guidelines.pdf

[32] http://www.depi.vic.gov.au/__data/assets/pdf_file/0017/251801/Habitat-Hectare-assessment-fact-sheet-May2014.pdf

[33] For details see http://www.dse.vic.gov.au/conservation-and-environment/native-vegetation-groups-for-victoria/vegetation-quality-assessment-manual

A1.2 Use of metrics in England

A1.2.1 Regulatory framework and offsetting requirements

There is currently no legal mandatory requirement for offsetting in the UK, beyond the Habitats Directive requirements[34] for compensatory measures for impacts on Natura 2000 sites in accordance with Article 6.4 and on strictly protected species in accordance with Article 12. However, the Countryside and Rights of Way Act (2000), updated by the Natural Environment and Rural Communities Act (2006), imposes what is known as the 'Biodiversity Duty' on certain public authorities –including local planning authorities – which requires that they must "have regard, so far as is consistent with the proper exercise of [their] functions, to the purpose of conserving biodiversity". This duty, along with associated requirements under the planning framework (see below) have resulted in some planning authorities seeking compensation for impacts on biodiversity, including the use of offsets (Treweek, 2009). Nevertheless, evidence for the Department of Environment, Food and Rural Affairs (Defra) suggests that its application has been sporadic with biodiversity impacts often not being taken into account (Tyldesley *et al*, 2012). Nevertheless, approximately 40% of local authorities have required some sort of offsetting or off site compensation at some point[35].

In 2011, the Government in effect adopted a 'No Net Loss' (NNL) policy with its publication of the Natural Environment White Paper[36] (HM Government, 2011) - albeit without establishing a time period by which to achieve it. The White Paper sets out to "move progressively from net biodiversity loss to net gain, by supporting healthy, well-functioning ecosystems and establishing more coherent ecological networks." The White Paper builds on the findings of a review of England's wildlife sites and ecological network (Lawton *et al*, 2010), which concluded that the Government should improve the quality of current wildlife sites by better management; increase the size of existing wildlife sites; enhance connections between sites; create new sites; and reduce pressures on wildlife by improving the wider environment.

Following the change in Government in 2010, planning policy has been subject to review and a significant reform, which resulted in the publication of the National Planning Policy Framework (NPPF)[37] by the Department for Local Communities and Government in March 2012. The NPPF reflects the Natural Environment White Paper by stating that the achievement of sustainable development includes 'moving from a net loss of bio-diversity to achieving net gains for nature' (para 9) and that 'the planning system should contribute to…minimising impacts on biodiversity and providing net gains in biodiversity where possible' (para 118). Furthermore, paragraph 118 also states that "When determining planning applications, local planning authorities should aim to conserve and enhance biodiversity by applying the following principles: if significant harm resulting from a development cannot be avoided (through locating on an alternative site with less harmful impacts), adequately mitigated, or, as a last resort, compensated for [emphasis added], then planning permission should be refused;". It also requires the mapping of local ecological networks.

It may therefore be argued that NNL is in effect mandatory, and that offsetting is also mandatory for significant residual impacts. Some local authorities are therefore requiring developers to achieve NNL and carry out offsetting where necessary according to this policy, such as in Warwickshire[38] (which is a pilot offset area – see below). However, most others are not, as Local Plans, which are the primary consideration in planning decisions, need to provide the legal

[34] As transposed in the UK through The Conservation (Natural Habitats, &c) Regulations 1994.

[35] Policy Exchange (2012) *Nurturing Nature*

[36] A White Paper produced by the UK Government sets out details of future policy on a particular subject and is often the basis for a Bill to be put before Parliament. The White Paper allows the Government an opportunity to gather feedback before it formally presents the policies as a Bill.

[37] https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/6077/2116950.pdf

[38] http://www.cieem.net/data/files/Resource_Library/Conferences/2014_Spring_Bio_Offsetting/09_David_Lowe.pdf

requirement for offsetting. As the NPPF is relatively new most Local Plans are still be revised and are not therefore necessarily NPPF-compliant yet. Uncertainty over what is meant by 'significant harm' is another factor that appears to be reducing requirements for offsets.

Where offsetting is carried out that is not connected to the Habitats Regulation, then Section 106 agreements under the Town and Country Planning Act 1990 are often used as delivery mechanisms. These allow a local Planning Authority to enter into a legally binding agreement or planning obligation with a landowner in response to the granting of planning permission.

To help move towards net gain of biodiversity and respond fulfil the requirements of the Lawton Review Defra have promoted a new voluntary approach for offsetting (the principles of which are outlined in Box 6) and instigated pilot trials in partnership with local authorities in six selected areas in England. The pilots were established in 2012 in Devon; Doncaster; Essex; Greater Norwich; Nottinghamshire and Warwickshire, Coventry and Solihull). Despite encouragement from the Government and assurances that the scheme should not be an additional burden to businesses, the uptake of pilot offsets by developers has been disappointing, with no developments expected to use offsets within the two-year pilot period (Evans, 2013). The formal pilot period is now over, and the pilots' achievements are currently being evaluated with a report expected to be published by Defra in July 2014.

Box 15 The principles that guided Defra's proposed approach to biodiversity offsetting

Offsetting should:

- Deliver real benefits for biodiversity by:
    - seeking to improve the effectiveness of managing compensation for biodiversity loss;
    - expanding and restoring habitats, not merely protecting the extent and condition of what is already there;
    - using offsets to contribute to enhancing England's ecological network by creating more; bigger, better and joined areas for biodiversity (as discussed in Making Space for Nature)
    - providing additionality; ie not being used to deliver something that would have happened anyway;
    - creating habitat which lasts in perpetuity;
    - being at the bottom of the mitigation hierarchy, and requiring avoidance and mitigation of impacts to take place first.
- Be managed at the local level as far as possible:
    - within national priorities for managing England's biodiversity;
    - within a standard framework, which provides a level of consistency for all involved;
    - through partnerships at a level that makes sense spatially, such as county level, catchment or natural area;
    - with the right level of national support and guidance to build capacity where it is needed;
    - involving local communities.
- Be as simple and straightforward as possible, for developers, local authorities and

others.

- Be transparent, giving clarity on how the offset calculations are derived and allowing people to see how offset resources are being used.
- Be good value for money.

In September 2013, Defra published a Green Paper[39] for consultation on the options for an offsetting scheme in England, in which it states the Government's preference for a voluntary approach, allowing developers to opt-in to the use of offsets in order to fulfil their requirements under the NPPF (DEFRA, 2013). An inquiry into the Green Paper was undertaken by the House of Commons Environmental Audit Committee (2013) which reported in 2013 and raised concerns on a number of issues including the need to adhere to the mitigation hierarchy, requirements for local offsetting and the simplicity of the metric (as further discussed below). It also noted that a mandatory offsetting system would encourage a market to develop, which would allow more environmentally and economically viable offset projects to be brought forward. It therefore recommended that the Government should allow the offset pilots to run their course and be evaluated before further developing proposals for offsetting. This has now been accepted by the government and further proposals are not expected before autumn 2014.

A1.2.2 Types of metric used and examples

Recommended metric used by pilot offsets

A form of habitat area x value x condition metric was developed for use in pilot offsets in England (DEFRA and Natural England, 2012), drawing on previous commissioned studies (Temple *et al*, 2010; Treweek *et al*, 2006). The metric firstly takes into account the inherent importance of each habitat and the condition of each habitat at the site. Habitats are assigned to one of three bands of distinctiveness, which broadly equates to inherent value (or uniqueness as assessed in some German metrics – see A1.4.2). National distinctiveness bands were set through expert consultations, taking into account parameters such as species richness, diversity, rarity (at local, regional, national and international scales) and the degree to which a habitat supports species rarely found in other habitats.

Habitats listed in the Biodiversity Action Plan are in the top band and are allocated a score of 6. Other semi-natural habitats are in the second band and score 4. Artificial habitats, such as intensive farmland, are in the lowest band and score 2. It is noteworthy that the weighting for natural and semi-natural habitats compared to artificial habitats is relatively modest.

Under the scheme, offsetting should ideally be like-for-like in the top band, and certainly within the same band; within the same band or through trading up for the second band, and through trading up when habitats in the third band are impacted. It is recognised that in some cases authorities may wish to modify the distinctiveness assessments according to local circumstances (for example if a habitat of moderate importance is particularly rare in the local area).

The condition of sites is assessed according to three bands, using a condition assessment tool developed for agri-environment schemes which is described in the Farm Environment Plan Handbook[40]. This is a standardised but relatively simple condition assessment methodology. The condition bands are scored, which when multiplied by the habitat distinctiveness scores results in the product scores for each combination shown in Table A1.1.

[39] Green Papers are consultation documents produced by the UK Government, normally when a government department is considering introducing a new law. The aim of the document is to allow people both inside and outside Parliament to debate the subject and give the department feedback on its suggestions.

[40] http://publications.naturalengland.org.uk/publication/32037

Table A1.1 Matrix showing how condition and distinctiveness are combined to give the number of biodiversity units per hectare, using England pilot offset metric

		Habitat distinctiveness (ie value)		
		Low (2)	Medium (4)	High (6)
Condition	Good (3)	6	12	18
	Moderate (2)	4	8	12
	Poor (1)	2	4	6

Source: (DEFRA & Natural England, 2012; Parkes et al, 2003).

The area of the habitat impacted is then multiplied by its combined score. Thus for example, 10 ha with the highest score would have a total score of 180 debits. The offset credits would then need to exceed this to achieve no net loss.

The primary score is then adjusted through the following secondary multipliers:

- Difficulty of restoring a site:
  - Low = 1
  - Medium = 1.5
  - High = 3
  - Very High = 10
- Spatial issues:
  - Offset is in allocation identified in the offsetting strategy (ie in a priority location for contributing to ecological networks) = no multiplier required (=1).
  - Offset is buffering, linking, restoring or expanding a habitat outside an area identified in the offsetting strategy = 2.
  - Offset is not making a contribution to the offsetting strategy.
- Time gaps between impact and expected offset provision: based on a discount rate of 3.5%, resulting in for example a multiplier of 2 for a gap of 20 years.

The Environmental Audit Committee (2013) revealed that the simplicity of the metric is welcomed by developers because it is relatively transparent and easy to interpret and does not require lengthy and costly surveys. However, conservation organisations and ecological scientists have criticised it for being too simplistic, in particular regarding the low number of distinctiveness and condition categories, its incompleteness regarding assessment of some biodiversity attributes (such as ecological networks) and the need for value judgements to be made (which are likely to vary greatly amongst surveyors).

Some also suggest that species should be included in the metric although Defra had given the following two reasons for not including them:

- A guiding principle is that there are to be no changes to existing levels of protection for biodiversity: therefore species protected under European legislation are excluded from the scheme.
- The metric should be universally usable: many of the species are localised and different species would be important in different areas, requiring a significant degree of local interpretation.

However, a meeting of ecological experts organised by the Centre for Ecology and Hydrology noted a number of reasons why assessing offset requirements according to habitat distinctiveness and condition alone would not safeguard species (see Table A1.2).

Table A1.2 Reasons why the size, distinctiveness and condition of each of the habitats present on a development site is not a reliable basis for safeguarding individual species in biodiversity offsets

Reason	Explanation / example
Habitat structure (the arrangement of features such as soil layers and types of plants) affects the occurrence of species.	See, for example, Hewson *et al*. (2011). Structure could be addressed through habitat condition scoring, although it is vital that critical information is not lost in the process of calculating offsets
Some species depend on a mix of different land cover types, vegetation types and landforms.	This dependence occurs among both migratory species and non-migratory species. Studies of the association between habitat and species can help to identify cases where this applies.
The management of a habitat affects the mix of species present and their abundance.	Densities of farmland birds depend on particular management practices.
The existence of predators, pests and diseases affects the distribution of individual species.	For example, domestic cats associated with housing development can deter birds and other animals.
The presence or absence of a species at a particular site is often determined by the interaction of a number of factors.	Some of these factors are mentioned above. Others may include the management history of the site or its interconnectedness with other sites that host the species.

Source: Based on Howard et al (2013)

For these, and other, reasons the CEH meeting concluded that if offsetting is to contribute towards a goal of ‘no net loss’ of biodiversity, assessment of the impacts of a development it must take into account the abundance of individual species in the wider landscape within which it is situated. Therefore they recommend that an expert-led assessment process is initiated to identify Species of Principal Importance for which habitat is not a suitable proxy indication of their presence or absence.

Metric used by Somerset County Council

Somerset County Council, has developed a species-led metric which concentrates on the requirements to maintain species’ populations in Favourable Conservation Status (Somerset County Council, 2014). The rationale for this is that protected species and other important species in the wider countryside are more likely to be affected by development than important habitats, because the latter are mostly contained within protected areas or local wildlife sites. Therefore to provide a more robust approach to ensuring no net loss of biodiversity the species metric aims to complement the Defra habitat metric (described above). The general habitat metric is used in Somerset as normal, but the species metric is also used for each species that cannot be reliably assessed by the general habitat metric.

The species metric has already been used in relation to Appropriate Assessments but the County Council[41], has developed a Biodiversity Offsetting methodology for use in the wider planning

[41] http://www.somerset.gov.uk/policies-and-plans/strategies/biodiversity-offsetting/

system for significant impacts on species of Community interest and nationally protected and threatened species wherever they occur.

The metric is based on a proposal by Temple *et al* (2010), and results in outcomes that are rated from 0 to 18, as in the Defra metric. This enables nesting of offset requirements produced by the two metrics. Requirements are not added (to avoid double-counting), but instead each species habitat and general habitat type requirements are compared and the greatest requirement for each habitat type taken as the final offset requirement are compared.

The metric is based on the Habitat Evaluation Procedures (HEP) developed in the USA (see section 3), which calculate Habitats Units (HU) based on the product of the area of each type of habitat affected multiplied by the habitat's suitability estimated as a Habitat Suitability Index (HSI) for each species. Figure A1.1 outlines the data requirements for the metric and how it is used in planning decision making.

Figure A1.1 Diagram showing data inputs and analytical stages in the calculation of the biodiversity metric used in Somerset, and its use in the planning process

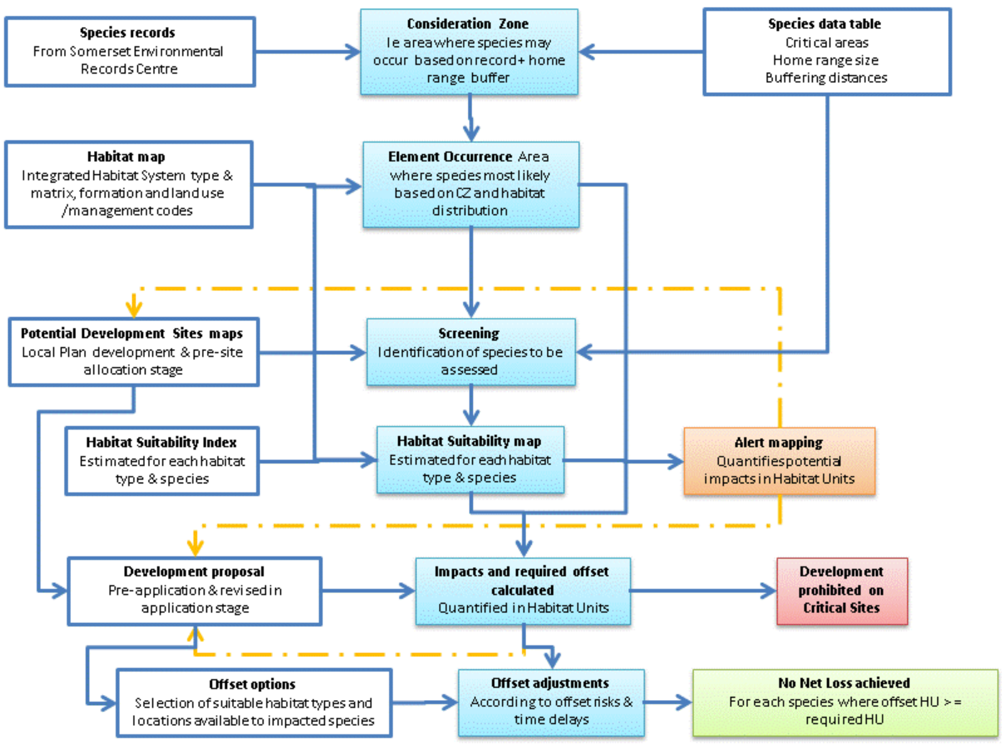

Table A1.3 below provides a hypothetical slightly simplified worked example of the metric, with each of the inputs and calculations further described below. The table is completed for each individual important species (identified within the screening process) or an umbrella species for species that have very similar habitat requirements. The umbrella species for a habitat is that which is considered to be most sensitive to the impacts of the development, and will therefore require the largest / highest quality offset.

The tables (based on an Excel worksheet tool) provide information for each defined field or sub compartment unit in each row, the area of which is included in column B. The Integrated Habitat System (HIS) type and code is indicated for each field /compartment. The IHS has over 400 habitat categories which integrates existing classifications in use in the UK, including those listed in Annex 1 of the habitats Directive and classifications developed as part of the national Biodiversity Action Planning process.

Sheet 1: calculation of lost units

Column D indicates whether the area would still be accessible to the species following the development, eg taking into account barriers to movement. If it is not available then it is counted as lost habitat and therefore contributes to column L, even if it is retained (ie not destroyed or degraded by the development).

The HSI score is taken from a list for each species for IHS habitat type based on a methodology developed by the USFWS (described in the USA section below). The HSI score is derived from a review of literature and expert judgements, verified where possible through comparisons with actual species distribution data. The HSI score ranges from 0 for habitat of no value to the species to 6 for excellent habitat that provides the species needs in total or provides an integral part of its lifecycle without which would not be able to maintain its existence. The HSI score for each HSI type, is modified according to additional habitat information where available on the habitat matrix, formation and land use / management. These additional habitat characters make the habitat more or less suitable for a species (see columns G-I) producing an adjusted HSI score (column J).

Column K then takes into account the density of the species within the part of the consideration zone that falls within the field / compartment. The expected density is based on circular band widths originating from the spatial location of each species record, which vary from species to species depending on its typical home range size. The band widths for each species are estimated from the literature and stored in the species data table (see figure A1.3) but may be modified for each calculation on the basis of field surveys carried out as part of the development's ecological impact assessment. Three band widths are defined that are given a score of 1-3. Thus the combination of HSI scores from 1-6 and band density scores from 1-3 give a range of per hectare Habitat Units (HU) ranging from 1 – 18 (as in the Defra habitat metric).

The number of lost HU as a result of residual impacts from the development in the example below is 63.66 HU (i.e. the total of all units lost or unavailable across all fields).

The most appropriate habitat to be restored to offset the residual impacts is considered to be lowland meadow. The required area of this habitat type for the offset is therefore adjusted by three multipliers relating to the risk that the offset habitat will not be adequately restored / created, that it may not be spatially adequate (eg it might become isolated from the impacted population) and that there will be a time delay before it is functional.

Sheet 2: Calculation of retained or enhanced Habitat Units

Sheet 2 includes each of the fields included in Sheet 1 but calculates the Habitat Units for those that would be retained and/or enhanced AND that would be available to the species (marked 'Y' in Sheet 1 column D). The units are calculated in the same way based on the HSI score and its modifiers for each future habitat type. The potential net gain in Habitat Units is then calculated by subtracting the existing values (listed in column D) from the predicted Habitat Units existing once the development is completed (column M). Thus the potential on-site gain is 19.48 and this can therefore be subtracted from Habitats Units required from the offset calculated in Sheet 1 (i.e. 114.6) giving a net offset requirement of 95.1 Habitat Units.

Table A1.3 Hypothetical examples of the calculation of offset requirements for a bat species in relation to a development using the Somerset version of the Habitat Evaluation Procedures metric

Sheet 1: Calculation of lost Habitat Units resulting from the development (see text for key and explanation)

A	B	C	D	E	F	G	H	I	J	K	L	M
Field / Compartment No.	**Area (ha)**	**Current habitat**	**Available for future use by species' population?**	**IHS Codes**	**HSI Habitat Score**	**HSI Matrix Score**	**HSI Formation Score**	**HSI Management / Land Use Score**	**Adjusted HSI Score**	**Consideration Zone**	**HU Lost**	**HU Retained, Accessible and to be Enhanced**
3	3.683	Arable	N	N/A	0	0	0	0	0	3	0.00	0.00
5a	2.2865	Arable	N	CR0.CL1	1	1	1	0.2	0.2	3	1.37	0.00
5b	0.3685	Arable	Y	CR0.CL1	1	1	1	0.2	0.2	3	0.00	0.22
6	4.561	Arable	N	CR0.CL1	1	1	1	0.2	0.2	3	2.73	0.00
8	3.253	Arable	N	CR0	1	1	1	0.2	0.2	3	1.95	0.00
10	7.7	Semi Improved Grassland	N	GU1	2	1	1	1	2	3	46.20	0.00
13	6.325	Arable	N	CR0.CL1	1	1	1	0.2	0.2	3	3.79	0.00
14a	11.828	Arable	N	CR0.CL1	1	1	1	0.2	0.2	3	7.10	0.0
15	1.869	Arable	Y	CR0.CL1	1	1.2	1	0.2	0.24	3	0.00	1.34
P2	0.014	Pond	N	AS0, AP1Z	3	0.75	1	1	2.25	3	0.094	0.00
H27	0.048	Hedgerow	Y	LF11, LM2	3	1	1	1	3	3	0.43	0.00
H33	0.0354	Hedgerow	Y	LF11, LM2	3	1	1	1	3	3	0.00	0.01
H49	0.048	Hedgerow	N	NA	0	0	0	0	0	3	0.00	0.00
TOTAL											**63.66**	

Required offset	Multipliers		
Type	**Spatial risk**	**Delivery risk**	**Temporal risk**
Lowland Meadow	1	1.5	1.2
Adjusted total HU required			**114.6**

Sheet 2: Calculation of retained or enhanced on-site Habitat Units resulting from the development (see text for key and explanation)

A Field / Compartment No.	B Area (ha)	C Current habitat	D HU Retained, Accessible and to be Enhanced	E Future habitat / land use	F Future IHS Codes	G HSI Habitat Score	H HSI Matrix Score	I HSI Formation Score	J HSI Management / Land Use Score	K HSI Score	L Consideration Zone	M Future HU	N Net gain on site (HU)
3	3.683	Arable	0.00	Road / Housing						0	3	0	0.00
5a	2.2865	Arable	0.00	Housing						0	3	0	0.00
5b	0.3685	Arable	0.22	Meadow with hedgerow	GN1,	3	1	1	1	3	3	3.31	3.09
6	4.561	Arable	0.00	Housing						0	3	0	0.00
8	3.253	Arable	0.00	Housing						0	3	0	0.00
10	7.7	Semi Improved Grasslland	0.00	Mixed use						0	3	0	0.00
13	6.325	Arable	0.00	Primary School						0	3	0	0.00
14a	11.828	Arable	0.0	Housing / Road						0	3	0	0.00
15	1.869	Arable	1.34	Meadow with hedgerow	GN1,	3	1	1	1	3	3	16.82	15.47
P2	0.014	Pond	0.00	Housing						0	3	0	0.00
H27	0.048	Hedgerow	0.00	Secondary School						0	3	0	0.00
H33	0.0354	Hedgerow	0.01	Hedgerow	LF111, LM32	4	1	1	1.5	6	3	0.64	0.63
H49	0.048	Hedgerow	0.00	Hedgerow	LF111, LM31	4	1	1	1.5	6	1	0.29	0.29
TOTAL													**19.48**

The metric requires detailed information on the ecology of protected species, their distributions and habitat maps. The cost of collating such information might not be justifiable if used for the metric alone, but it is important to note that the data allow assessments of potential biodiversity impacts, on habitats in general as well as species, in the early stages of the planning process. This early assessment can help avoid biodiversity impacts and, where offsets are necessary, ensure they are located correctly and in the appropriate time frame. Thus, strategic advance investment in biodiversity mapping and in ecological support within the authority, may smooth planning processes and thereby reduce the likelihood of delays and associated costs. Although data are not available to test this, given the high costs of project delays it seems plausible that investing in biodiversity data collation and GIS systems that enable early impact assessment may have overall economic benefits.

A1.2.3 Conclusions

The pilot scheme in England was voluntary and has not resulted in any offset schemes being agreed over the two-year trial period. However, the Defra metric has been used by all the pilots and has been studied and debated by others. Tyldesley *et al*. (2012) considered that the metric is valid and works, but noted that it does not assess impacted habitats in the context of their wider ecological setting. Ecologists and NGOs who gave evidence to the Environmental Audit Committee inquiry were more critical and therefore the committee concluded that “the biodiversity metric described in the Green Paper [i.e. that used in the pilots] is overly simplistic”. It went on to recommend that “If biodiversity offsetting is introduced, its metric for calculating environmental losses and gains must reflect the full complexity of habitats, including particular species, local habitat significance, ecosystem services provided and ‘ecosystem network’ connectivity.”

Furthermore as noted above, expert ecologists have recommended that particular species’ requirements should also be assessed alongside the metric where their requirements are unlikely to be accurately assessed by the habitat focussed metric (Howard *et al*, 2013). In this respect, the species-focussed approach developed by Somerset County Council and tested in planning decisions, does appear to be a practical and robust methodology. It requires a GIS with considerable data on the ecology and locations of important species and comprehensive and detailed habitat distribution data. But such habitat data are now increasingly available and it could be cost effective to strategically fill data gaps for identified species of principal importance for which general habitat metrics are not a suitable proxy. This is because good spatial biodiversity data allows the assessment of possible biodiversity impacts and potential offsetting costs to be considered at early stages in the planning process, which can therefore help developers avoid and reduce biodiversity impacts and associated costs.

A1.3 Use of metrics in France

A1.3.1 Regulatory framework and offsetting requirements

French law has included an obligation to offset unavoidable impacts on the environment since 1976, but offsetting was for the most part ignored or poorly applied until recently (Mathieu and Quétier, 2014, Annex in (Tucker *et al*, 2014). The French environment ministry published guidance on how to compensate for impacts on protected species and their habitats in 2012 (MEDDE, 2012a). In parallel to these changes, the Environmental Liability Directive was transposed in 2008[42]. There are on-going discussions to better define its scope through the introduction of third-party ecological damage (‘prejudice écologique’) in the French Civil Code.

[42] LOI n° 2008-757 du 1er août 2008 relative à la responsabilité environnementale et à diverses dispositions d'adaptation au droit communautaire dans le domaine de l'environnement (1) http://legifrance.gouv.fr/affichTexte.do?cidTexte=JORFTEXT000019277729, revised following the European Court of Justice ruling case C-241/08.

The EIA law was updated in 2010[43], with its associated decree which came into force in 2012[44], to require that project authorisations (permits) must describe offsetting measures for residual impacts and the corresponding monitoring, and to strengthen enforcement options (including penalties) to ensure that developers are liable for their offsets. Permitting authorities must monitor and verify that offsets are carried out (and the newly created French environment inspectors are responsible for monitoring compliance[45]). The French environment ministry published guidance on the mitigation hierarchy (MEDDE 2012b) and on how to implement avoidance, mitigation and offset measures (MEDDE 2013). This is forcing a reorganisation of the French permitting process as poorly designed projects face increased risks. In particular, the importance of biodiversity concerns in impact assessments has steadily increased, together with the more widespread use of sectorial procedures such as those targeting protected species and Natura 2000 sites, and is being investigated earlier in project cycles with more thorough application of the mitigation hierarchy. This contrasts with using offsets as a last minute solution to poor design, which remains widespread (Vanpeene-Bruhier *et al* 2013).

Mitigation of impacts for projects affecting water bodies are regulated in the Environment Code[46] and through the French River Basin Management Plans (known as SDAGE[47]). The plans were revised and updated in 2009, and several include a requirement for offsetting residual impacts from the destruction of wetlands through restoration of equivalent wetland functions and/or degraded wetlands.

The French Forestry Code[48] established that offsetting can be required for clearing of woodland, as part of the process of granting the felling license. Area-based multipliers are used to determine the area of new woodland to be planted to compensate for the clearing. This is done at the discretion of the permitting authorities as there are no guidelines on the circumstances in which an offset is required, and the multiplier to be used.

Scope: What plans or projects lead to offset assessments?

The EIA decree lists projects which are subject to the offset requirement. These include urban expansion, infrastructure, and industrial projects (e.g. energy, extractive industries). There are no offsetting requirements for residual impacts from agriculture, forestry or fisheries (Mathieu and Quétier 2014 Annex in (Tucker *et al*, 2014).

Strategic Environmental Assessment requirements for land-use plans also require offsets (‘compensation’). This was changed in the last few years (Quétier *et al*. 2014). Some recent revisions of land-use plans such as the Schémas de Cohérence Territoriale (SCOT) and Plans Locaux d’Urbanisme (PLU) have tried to plan for offsets but this work is largely on-going and few lessons can be learned from it.

In practice, offsetting in France has been restricted to projects affecting European and nationally protected species and their habitats, wetlands (as specified in river basin management plans), and some woodlands. A few large-scale transport projects have generated offsets that are sometimes considerable in area and cost (Vanpeene-Bruhier *et al* 2013), but biodiversity impacts of small projects and urban development by local governments tend to go unnoticed, in spite of considerable cumulative impacts.

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[43] Grenelle 2 law - LOI n° 2010-788 du 12 juillet 2010 portant engagement national pour l'environnement (1), http://www.legifrance.gouv.fr/affichTexte.do?cidTexte=JORFTEXT000022470434

[44] Décret n° 2011-2019 du 29 décembre 2011 portant réforme des études d'impact des projets de travaux, d'ouvrages ou d'aménagements, http://www.legifrance.gouv.fr/affichTexte.do?cidTexte=JORFTEXT000025054134

[45] police judiciaire du code de l'environnement created by Ordonnance n° 2012-34

[46] Code de l’Environnement Article R 214-6, http://www.legifrance.org/affichCodeArticle.do?cidTexte=LEGITEXT000006074220&idArticle=LEGIARTI000006835467&dateTexte=&categorieLien=cid

[47] SDAGE = Schéma Directeur de Gestion et d’Aménagement des Eaux

[48] Code Forestier 1 Juillet 2012 L341-6 http://www.legifrance.gouv.fr/affichCode.do?cidTexte=LEGITEXT000006071514&dateTexte=20120630

Scope: What impacts are assessed for possible offsetting?

The EIA law requires that offsets be applied to significant residual impacts on fauna, flora, and natural environments. Government guidance on how to implement offsetting requirements applies to natural environments, which are defined as terrestrial, aquatic and marine environments encompassing natural habitats, animal and plant species, features contributing to functional connectivity, and ecosystems, including their physical and biological components and the services they provide (MEDDE 2012b). This can include protected species and their habitats, and 'common' species and ecosystem functions, in particular hedgerows, unprotected wetlands, green spaces in urban areas and woodlands. Priority must be given to 'major issues', which include protected species and Natura 2000 sites, and key ecological connectivity features including for example migration areas, and key ecosystem services (MEDDE 2013). However, efforts are generally related to the level of conservation concern (e.g. species protection) or stakeholder pressure (e.g. some game species). Offsetting specifically for ecosystem functions is not currently operational in France (UICN France 2011).

Protected species: French law protects many national priority species as well as the species covered by the Habitats and Birds Directives. For some species, only individuals are protected, but not their habitat (Quétier *et al*, 2014).

Wetlands and floodplains: The French legal definition of a wetland[49] is very broad, such that even drained floodplain soils used for agriculture could be included (Quétier *et al*, 2014). As a result, offset requirements in river floodplains are quite extensive.

French guidance states that offsets should remain effective for as long as impacts last (MEDDE 2012b) but this is rarely enforced in practice, with permits requiring offset commitments for a few decades (to our knowledge the maximum duration is 52 years for the A65 motorway between Langon and Pau in South-West France[50]).

Scope: how are offset measures defined?

The 2010 EIA law specifies that offset measures must be in-kind and in "functional proximity" to impacts (i.e. but generally on the affected area or close by), and must aim to improve the overall environmental value[51]. The guidance also emphasises strictly targeted like-for-like equivalence at a local level (MEDDE 2012b). In practice, access to land is the determining factor so the location of offsets is opportunistic, and linked to the real estate market[52]. No guidance is available for any kind of 'like-for-better' or 'trading up' process where less valuable types of biodiversity can be offset with measures for higher priority biodiversity (Quétier *et al*, 2014). In practice, however, such trading-up is generalized as mitigation and offsets focus on more valuable components of biodiversity (e.g. rare protected species) to the detriment of less valuable components (e.g. common species).

A1.3.2 Types of metric used and examples

Only basic guidance is currently available in France on assessing ecological equivalence (MEDDE, 2012, MEDDE 2013). Offset requirements are determined on a case by case basis using a variety of methods (Quétier 2014 Annex in (Tucker *et al*, 2014). The government guidance recommends grouping components into broader categories (e.g. species sharing similar habitat requirements), with their associated metrics (e.g. the characteristics of the shared habitat) (Quétier *et al*, 2014).

[49] Ministerial Order DEVO0813942A of 2008
[50] Personal communication Francois Quétier, 6 May 2014
[51] Les mesures doivent être réalisées sur le site endommagé ou à sa proximité et doivent si possible améliorer la qualité environnementale des milieux.
[52] Personal communication, Fabien Quétier, 2 May 2014

Habitat area with multiplier

The French metrics currently in use are based on simple habitat area ratios with a choice of multiplier.

The wetland offset requirements in French river basin management plans (SDAGEs) are defined by simple area ratios with a multiplier. The destroyed wetland must be offset by recreation or restoration of an equivalent surface area within the same catchment, or if further away, by an area multiplied by a specified factor, for example using an area-based multiplier of 1.5 to 2 ha restored for 1 ha lost. Such requirements (with varying ratios) can be found in most management plans (Quétier *et al*, 2014). Wetlands are defined very broadly, such that even drained floodplain soils used for agriculture could be included (Quétier *et al*, 2014).

Box 16 Wetland offsetting in French river basin management plans

Some examples:

SDAGEs Loire-Bretagne and Rhône Méditerranée: Multiplier is either 1:1 or 1:2 of destroyed habitat area depending on the specified conditions of the offset. If the offset is within the same catchment area the multiplier is 1:1 (recreation or restoration in the same catchment area of a wetland with equivalent function and biodiversity). If the offset cannot take place within the catchment area or takes place further than 25 km from the wetland that is to be destroyed or if the catchment area is greater than 500km^2, and/or the optimal equivalence related to functions and biodiversity cannot be found, then the compensation should apply to an area equal to twice the size of the area which is being destroyed (1:2).

SDAGE Seine Normandie: Multiplier of 1:1 of destroyed habitat. Offsets must be in the same river basin.

SDAGE Adour Garonne: Creation or acquisition of wetland as offset can compensate a minimum of 150% of lost wetland area. Offsets must be at a 'coherent' scale.

Reference: Action 8-B2 in (Secretariát technique du basin Loire-Bretagne 2010), (Quétier *et al*, 2014), (MEDDE 2013)

The **Forestry Code** defines offsetting requirements for forest clearance (MAAF 2012). It specifies that compensation ratios for cleared forest areas should range from 1:1 (1 ha created to 1 ha destroyed) up to 1:5, depending on the ecological role and social significance of the affected area (MEDDE 2013). The afforestation can be in the same forest region or in a sector that is ecologically or socially comparable (Morandeau and Vilaysack, 2012). The choice of multiplier and the type of afforestation required is at the discretion of the permitting authorities[53] as there are no guidelines on the circumstances in which an offset is required, nor the multiplier to be used. The purpose of the legislative requirement is to ensure that felled areas are replaced with afforestation, and there is not necessarily any requirement for biodiversity conservation.

Species habitat metric

A more complex metric to aid developers in complying with the French protected species regulations[54] has been developed by the consultancy Biotope and used to calculate offsets for a large railway development in southern France. It focuses on offsetting residual impacts on protected species and their habitats, that remain after mitigation and restoration opportunities have been developed. The metric results in species-for-species offsets, on the basis of habitat quality for each species. The metric can work with a variety of exchange rules but by French law, offsets must be in the same area as the development project and effective within a reasonable

[53] DDT(M) (Departmental Territorial Directorate), with felling license awarded by the Préfet de département

[54] enforced by DREAL (Regional Directorate of the Environment, Development and Housing)

time frame. The method requires documentation of development project impacts, area affected, level of conservation concern for species, habitats quality, and maps thereof, and occurrence of species.

Box 17 Multi-Species-Habitat metric developed by Biotope

The metric focuses on species requirements, focusing on species with an unfavourable conservation status and a high conservation priority. The natural habitats in the study area are mapped and scored according to their suitability for the selected species and their importance for maintaining local population viability. The most suitable habitats are selected and defined in order to be representative of the species' needs and relevant with the way habitats are seen by the stakeholders.

Each species' habitat is compared to the habitat areas impacted by the development. Several species can use the same impacted habitat area; therefore the comparison is made using species clusters and habitat groups. Species are grouped according to their conservation status and habitats are grouped according to their broad natural environment type.

The group of species of highest conservation status (ie critically endangered) are assigned the highest coefficient (x3), endangered species are assigned a lower coefficient (x2), and the vulnerable species are assigned the lowest coefficient (x1). The habitats are also assigned coefficients according to their suitability for the species, ie very suitable (coefficient x3) down to not very suitable (coefficient x1). Coefficients can be adjusted but the hierarchy of conservation priorities has to be respected.

The residual impacted areas of the highest priority species are calculated first, and subtracted from the residual impacted areas of the lower priority species in a series of steps in order to obtain the net impacted areas. Each of these residual impact areas is multiplied by the appropriate coefficients for habitat suitability and species conservation status, to give an overall number of units per broad natural environment that needs to be offset.

The metric does not set multipliers as these result from the "gain per unit area" that differs between on-the-ground measures. The higher the benefits of the measure are, the lower the area needed to offset a given loss will be. Multipliers or a bonus/penalty system is used to take into account uncertainties and time-delays between impact losses and offset gains.

Reference: Fabien Quetier, Biotope Consultancy, Derogation demand for the CNM project (Contournement ferroviaire Nîmes – Montpellier)

A1.3.3 Use of multipliers in metrics

The government guidance (MEDDE 2013) recommends the use of a risk multiplier[55] to account for risks associated with:

- underestimate of residual impacts of project
- failure to achieve ecological value of offset measures due to natural factors
- time lag between impact and establishment of offset measures
- spatial separation between impact and offset measures and associated functional discontinuities

[55] In French: *coefficient d'ajustement*

- failure to achieve a net gain through the restoration of degraded habitats

The risk multiplier must be greater than a 1:1 area ratio of destroyed habitat to offset habitat, but no further guidance is given.

A1.3.4 Conclusions

The routinely used offset metrics in France are based on a simple habitat area with multipliers approach. For some difficult to restore habitats, such as peatlands, the area ratios are a very crude way to determine levels of acceptable loss of existing habitat in relation to the chance of upgrading existing habitat (Quétier and Lavorel, 2011). Wetlands, for example, vary greatly in the degree to which they can be successfully re-created or restored (Jähnig *et al*, 2011; Moreno-Mateos *et al*, 2012).

Developers are becoming concerned about increases in the ratios of offset area being required; for example up to 6 ha for each ha destroyed in the French region of Franche-Comté (Vanpeene-Bruhier *et al* 2013).

However, there is a great deal of activity on these issues and more sophisticated approaches are in discussion in several working groups and initiatives to build the professional skills of impact assessors and public authorities (Vanpeene-Bruhier *et al* 2013).

A1.4 Use of metrics in Germany

A1.4.1 Regulatory framework and offsetting requirements

Experience of offsetting in the EU has been most extensive and longstanding in Germany, because it has been a national mandatory requirement since 1976, when the Impact Mitigation Regulation (IMR) was adopted as part of the Federal Nature Conservation Act and the Federal Building Code[56]. Further, a wide range of offsetting approaches and metrics have been developed and used, which provides valuable comparative evidence of the various advantages and disadvantages. The German offsetting system and metrics are therefore described in some detail.

The German IMR requires firstly the avoidance of significant negative effects on natural assets and their functions[57], and secondly the mitigation of negative effects through specific measures[58], and thirdly the compensation of residual impacts. An indication of the volume of activity generated by this legal requirement is the estimate that in the federal state of Schleswig-Holstein, around 25,000 ha (1.6% of the state area) was allocated to offset measures by 2010 (Grundler & Thomas 2012).

German forest law[59], independently of the IMR requirement, protects forest areas and requires losses to be replaced with afforestation on an equivalent area. This afforestation can be considered as an offset under the IMR only if it includes measures that are additional to the restoration of the forest ecosystem services as defined in the forest law and good practice (although biodiversity and ecosystem service benefits may of course arise from the forest management goals).

German municipal land use planning under the Federal Building Code distinguishes between the zone defined for human settlement, which is subject to detailed spatial planning, and the rural

[56] The legal requirements are laid out in the Federal Nature Conservation Act (referred to as BNatSchG 1976 and BNatSchG 2009) Articles 13 to 19 and in the Federal Building Code (referred to as BauGB 1998 and BauGB 2004) Articles 1a, 13 and 13a.

[57] In German: *Gebot der Vermeidung von Eingriffen bzw. Beeinträchtigungen*

[58] In German: *Verminderung von Eingriffen bzw. Beeinträchtigungen*

[59] Referred to as BWaldG

zone in which developments are strictly limited except certain priority activities such as transport and energy infrastructure. Agricultural and forest areas should be included in rural zones. The IMR process is defined differently in the urban zone (to which the Federal Building Code applies) and in the rural zone (to which the Federal Nature Conservation Act applies).

The German Federal Nature Conservation Act that defines the impact mitigation regulations has been revised in 2002, 2007 and 2010 in attempts to improve the impact assessment and offsetting process. The latest revision in 2009-10 gave the German federal government the power to issue ordinances that define implementation details for offsetting in the rural zone (see below).

Scope: What plans or projects lead to offset assessments?

The key impacts covered by the IMR are associated with buildings, housing, transport infrastructure (roads, railways and waterways), energy infrastructure (power lines, wind-turbines, power stations etc.) and mining. Other project types may also fall under the scope of IMR according to the legal definition of an impact: "*Interventions in nature and landscape, as defined in this Act, shall refer to any changes affecting the shape or use of areas, or changes in the groundwater level associated with the active soil layer, which may significantly impair the performance and functioning of the natural balance or landscape appearance*" (translation of Federal Nature Conservation Act).

Certain developments are exempt from the offset requirement:

- In the rural zone, the impacts of agriculture, forestry and fishing are exempt so long as they follow good practice standards (such as cross-compliance rules or sustainable forest management).
- In the urban zone, housing developments within a completely urban area that do not include any green space and do not exceed 2 ha are exempt from environmental assessment and offsetting requirements; developments that exceed 2 ha but do not exceed 7 ha require a screening assessment to determine whether a full EIA (and possibly offsets) is necessary or not.[60]

Scope: What impacts are assessed for possible offsetting?

The law requires mitigation and offsetting of impacts on both biodiversity (natural assets) and environmental functions, and their interaction with human wellbeing – ie ecosystem services. Natural assets and their functions include: fauna and flora and their interactions including biodiversity as a whole[61], the aesthetic and recreational quality of the landscape, soil, water, climate and air quality, and interactions between these[62]. The assessment should focus on significant known impacts, and does not have to consider all possible impacts.

German law defines the species protected in the Birds Directive Annex 1 and Habitats Directive Annex IV [63] as strictly protected species. Impacts on these species and impacts on the Natura 2000 network must be assessed separately following the appropriate assessment procedure. The criteria and methods for determining offset requirements for these species of European concern must follow the European Commission guidance (European Commission, 2007) and are defined in German law[64] accordingly (Runge *et al* 2010).The methods for assessing offset requirements for European protected species are therefore not discussed further here.

The law requires that impacts on the following natural assets are assessed for possible offsetting:

[60] Specified in the revision of the Federal Building Code in 2007 (BauGB Novelle 2007)

[61] In German: *Biologisches Wirkungsgefüge*

[62] Defined in BauGB §1 Abs. 6 Nr. 7 and BauGB §1a

[63] Strictly protected species (*streng geschützte Arten*) in BNatSchG §10 paragraph 2 articles 10 & 11; BNatSchG §7 paragraph 2 articles 13 & 14

[64] Defined in § 44 Abs. 1 and Abs. 5 BNatSchG and § 45 Abs.8 BNatSchG

- **Species**: In addition to the strictly protected species, German law defines a range of species as specially protected species[65], including all native bird species, reptiles and amphibians, most mammal species, and many insects and plants. There is currently no ordinance detailing how these species are to be treated in the impact assessment process, but the national conservation agency has published lists of species for which Germany has a national responsibility[66]. Some federal states have issued their own guidance to which species are subject to individual population/habitat protection in the planning process (eg colony breeders), and which species should be evaluated as functional groups (Runge *et al*, 2010). In practice, planning authorities generally consider impacts on other national and regional priority species in the offset rationale (Busse *et al* 2013).
- **Habitats/biotopes**: The impact mitigation regulation does not refer directly to habitats, but in practice evaluations are done on the basis of broad habitat types or biotopes[67]. Most assessment methods require the production of a map dividing the affected area into biotope types, and all states have produced a list of standard biotope types in their region. Particular biotope types are legally protected in some federal states (for example hedge-banks in Schleswig-Holstein, ponds and other small water bodies and hedge banks in Niedersachsen[68]).
- **Landscape**: The law specifies the assessment of the landscape's aesthetic quality and recreational functions.
- **Soil functions**: The soil assessment includes the loss of the productive function of soil in relation to agricultural crops or natural vegetation, the significance of the soil's role as buffer and filter for pollution, the resistance of soil to erosion, for regulating water flows and replenishing groundwater stocks, as a habitat for species, and as a historical site e.g. of archaeological interest. The area affected by soil sealing is quantified and must be compensated with an equivalent or larger area that is either unsealed or restored sufficiently to compensate for the lost soil functions, e.g. through the conversion of a sufficiently large area of arable soil to permanent grassland or other vegetation.
- **Water functions**: The water assessment includes impacts on surface water bodies both through direct loss of habitat and indirect impacts, and impacts on groundwater replenishment, storage capacity and quality. The groundwater impacts are generally directly related to the area of soil sealed.
- **Air and climate functions**: The assessment includes the impact of an increase in impervious surfaces on the local climate in built up areas (e.g. loss of capacity for cold air creation[69]).

The law also requires an assessment of impacts on interactions between functions and assets, but this is generally interpreted as being subsumed within the assessment of impacts on biotopes[70].

Scope: What area is assessed for impacts and possible offsetting?

The German IMR applies to all affected areas, independently of their biodiversity value, and the assessment methodology must measure impacts on the ecosystem and landscape across the whole assessed area. The total assessed area includes both the directly impacted area (e.g. the building footprint) and its surroundings (referred to here as the development area). Theoretically,

[65] Specially protected species (*Besonders geschützte Arten)* in BNatSchG §

[66] http://www.bfn.de/0302_verantwortungsarten.html

[67] These are referred to as biotopes following the German language usage, as the German word 'habitat' refers specifically to the habitat of a species.

[68] Defined in § 28a and § 33 Niedersächsisches Naturschutzgesetz

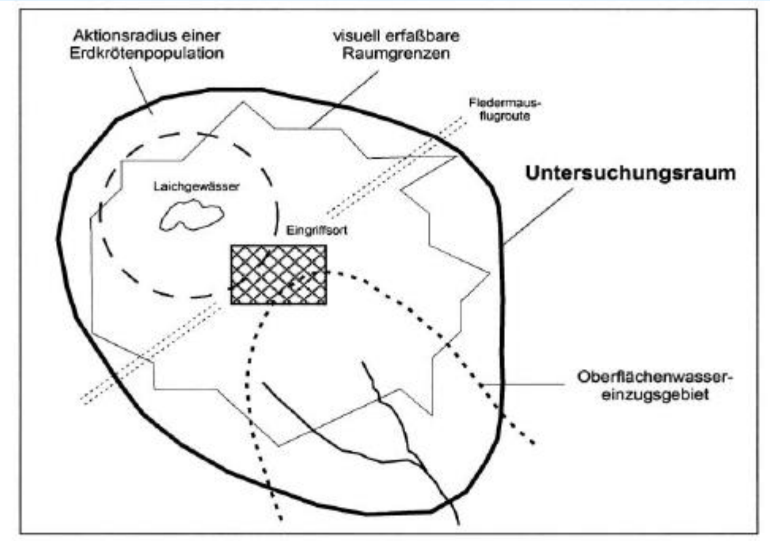
[69] In German: *Kaltluftentstehungsgebieten*

[70] See Zu § 5 in BKompV Begründung besonderer Teil draft April 2013 and § 7 paragraph 2 no 4 BNatSchG 2009

the total assessed area should be defined according to the extent of expected impacts; however in practice it cannot be ruled out that local authorities determine the impact area based on land planning parcel boundaries. Some federal states provide explicit guidance on how to define the total assessed areas; others leave the question open.

Box 18 How to define the impact area

In Sachsen, clear guidance is provided on how to define the impact area (SMUL 2009a, 2009b). The impact area should be defined case-by-case according to a reasonable estimate[71] of the extent of the directly impacted area, the indirectly affected area[72], and including the offset area[73]. The impact area should be defined as soon as possible in agreement between developer and planning authority, and should be adjusted during the impact assessment if new information comes to light. It is also necessary to define at this stage whether protected species are affected, and/or whether the assessment of biotope impacts requires a key indicator species survey in the field. Also to decide whether the impact assessment of ecological functions requires specific field data other than that provided by the biotope mapping.



Mapping impact area according to ecological functions (SMUL 2009a)

References: (SMUL 2009a, 2009b), http://www.umwelt.sachsen.de/umwelt/natur/8516.htm

Scope: How are offset measures defined?

Type of offset: The German impact mitigation regulation differentiates between offset measures that provide a strictly 'like-for-like' or 'in-kind' offset[74] (known as restoration measures - *Ausgleichsmassnahmen*), and offset measures that provide the same value but not necessarily exactly the same type of habitat or function ie 'like-for-unlike' or 'out-of-kind'[75] offsets as well as 'like-for-like' (known as replacement compensation - *Ersatzmassnahmen*). The 2002 revision of the Federal Nature Conservation Act loosened the prioritisation of *Ausgleichsmassnahmen* and the close spatial and functional connection between impact and offset, opening up the opportunity to use habitat banking systems (Wende *et al*, 2005). The 2009 revision of the Federal

[71] Legal principle of proportionality (in German = *Verhältnismässigkeit*)

[72] In German: Eingriffsraum

[73] In German: Kompensationsraum

[74] The German law uses the word '*gleichartig*' ie same or similar in characteristics

[75] The German law uses the word '*gleichwertig*' ie the same value

Nature Conservation Act gives the two types of measure the same legal priority in the rural zone, as is already the case under the Building Code. Offsets must be within the same bio-geographical region[76] as the development they are offsetting. This allows a more flexible approach to finding appropriate offsets, particularly in connection with the use of compensation pools or habitat banking. However, there are also fears that this legal revision is further weakening the connection between offsets and damage (Breuer 2010). The rationale for offsets that replace affected functions supposes that they must usually be in close physical proximity to the impact, particularly with regard to landscape aspects[77]. In practice, offset measures in the settlement zone (ie where the local authority is the planning authority) are usually located within the same local authority area. The draft federal offsetting ordinance published in April 2013 provides examples of like-for-like and like-for-unlike offset measures.

Compensation payments: The German law allows for compensation payments in cases where on-the-ground offsets are not possible. During the 2010 revision there was considerable political pressure to make compensation payments equivalent to offset measures[78]. This was successfully resisted by various parties, arguing that offset measures can be more cost-effective than payments (BFAD 2011), and that removing the requirement to implement on-the-ground measures wherever possible destroys the link between damage and compensation (DRL 2007, Breuer 2010). The draft federal ordinance published in April 2013 declared that compensation payments can be required for impacts that are impossible to offset physically, notably the landscape impacts of energy infrastructure such as high voltage power lines and towers and wind turbines over 20m in height, but also buildings, excavations, deposits. The payments should be calculated on the basis of number of towers and height/height plus blade length. The draft federal ordinance also defines how compensation payments should be calculated, as this is currently regulated differently in each federal state. For example, Thüringen specifies that the cost of offsets should not exceed 10% of total development costs (Freistaat Thüringen 2005).

The German EIA Association criticises the prioritisation of monetary compensation for impacts of energy infrastructure on landscape scenery (UVP-Gesellschaft 2012). They argue that it is actually easy to compensate for landscape scenery impacts by removing old electricity infrastructure facilities or any other abandoned infrastructure. Thus, this should lead to a prioritisation of actual compensation or mitigation measures instead of the monetary approach.

Offset measures in protected areas: The German law allows offset measures to include restoration and habitat creation in protected areas and measures defined by Natura 2000 management plans or river basin management plans, provided they adequately offset the impacts. This corresponds to a political desire to use the offset rules to achieve wider nature conservation goals, including connectivity of the Natura 2000 network, measures aiming at achieving favourable conservation status of species protected by the Habitats and Birds Directives, and continued ecological functionality measures (Breuer 2010)[79]. Although the law excludes measures that must be carried out because of legal obligations[80], it does not further define this with respect to activities related to the Habitats Directive or the Water Framework Directive goals. A review in 2006 found that compensation pools were funding some offset measures that are actually government responsibilities (Thum 2006). The German NGO NABU has criticised the law and the draft ordinance for encouraging the use of offsets to replace public funding for restoration and maintenance measures[81]. In practice some habitat bank operators

[76] Germany is divided into 49 such regions or '*Naturräume*' (described in BKompV draft 2013 Annex 2)

[77] As defined by a ruling of the German high court (BVerwG) of 27 Sept 1990 (Case 4C 44.87)

[78] For example from the Niedersachsen government (Breuer 2010) and the Hessen Ministry of Commerce (BFAD 2011)

[79] See draft federal ordinance § 2 paragraph 4 BKompV. See also the Hessen Ordinance on the execution of compensation measures, eco-accounts, their negotiability and the setting of compensation charges § 2 Section 1 no 2 / https://umweltministerium.hessen.de/umwelt-natur/naturschutz/eingriff-kompensation/oekopunkte

[80] Defined in § 15 paragraph 6 BNatSchG 2009

[81] Stellungnahme des NABU zum Kabinettsbeschluss einer Kompensationsverordnung des Bundes. NABU 7 May 2013. http://www.nabu.de/themen/naturschutz/naturschutzrecht/news/15267.html

state how difficult and cost intensive it is to generate sufficient eco-points from offset measures in protected areas, due to the high baseline value of their habitats[82]. Thus, in practice it is actually not preferred to use this legal opportunity.

Offset measures on agricultural or forest land: There is political pressure to restrict the conversion of productive agricultural and forest land into offsets. There is a legal duty on planners to avoid the conversion of productive land wherever possible, and the draft federal ordinance of April 2013 provides some guidance on how to do this. According to the federal draft, offset measures should only take land out of productive use for agriculture or forestry if it is not possible to offset using measures that remove artificial surfaces (unseal soil), improve connectivity, or that upgrade the biodiversity value of agricultural or forestry land through restoration and management measures[83]. Offset measures can include production-integrated measures such as low-input agricultural production systems; however, this presents several challenges to the ecological and legal credibility of offsets and may be controversial (Busse *et al* 2013). The draft federal ordinance provides a procedure for defining productive agricultural land and for how to consult agricultural and forestry authorities with regard to planning offsets on agricultural or forest land. It recommends that the regional landscape plan should define suitable areas so that there is no need for a consultation on every offset plan.

A1.4.2 Types of metric used and examples

Diversity of offset metrics in Germany

The German regulation requires the planning authority to balance the unavoidable negative effects with specific mitigation and offsetting measures. However the federal law does not specify what processes or metrics should be used to evaluate and weigh up the negative impacts, and also leaves a number of key terms undefined (Darbi and Tausch, 2010). The planning authorities have developed methods 'from the bottom up', and at least 40 published approaches exist (Bruns, 2007).

The 16 German federal states have attempted to reduce the resulting legal uncertainty for local planning authorities by producing state level guidance. However, some local authorities continue to follow guidance from other federal states, or use hybrid methods, or have developed their own (e.g. Kreis Recklingshausen 2013). For example, in one district in the federal state Baden-Württemberg, five local authorities were found to use the Baden-Württemberg model, whilst 14 were using the Bavarian model, and several others were using models from the federal states Hessen and Rheinland-Pfalz (Mazza & Schiller 2014). Some German local authorities have produced their own guidance.

The most widely and most often used metrics are those developed by federal states with the highest frequency of development projects in the last 25 years, namely Baden-Württemberg, Bayern, Brandenburg, Nordrhein-Westfalen, Rheinland-Pfalz, Sachsen and Thüringen.

German companies and other developers are particularly critical of the offsetting rules when they plan large infrastructure projects that cross federal state borders, and are obliged to deliver two sets of offset measures for similar impacts that may differ considerably in price between federal states, due to differences in metrics[84]. Companies setting up onshore wind farms have also highlighted the differences in offset requirements in different federal states. The regional nature conservation authorities responsible for offsetting in the rural zone therefore often specify in the planning application that the offset calculation must use federal-level guidance developed specifically for that type of development. Such federal-level metrics exist for railways (EBA 2010) and waterways (BFG 2010). State-level guidance for road building is provided in Mecklenburg-

[82] e.g. Jörg Voss of Saxony site agency, oral statement on 16th of May 2013 at No Net Loss Stakeholder Workshop in Germany

[83] See § 10 BKompV draft April 2013

[84] Personal communication, Elke Bruns 1 May 2014

Vorpommern (Froelich & Sporbeck 2002), Brandenburg (MIR 2009), and others; for electricity infrastructure by Niedersachsen (NLT 2011) and others.

Various attempts have been made to unify offset metrics and methods across Germany, particularly after the offset process was integrated into the urban planning process in the Building Code in 1998, which conservation groups feared would result in a lower standard (Bruns, 2007). A working group of the conference of German environment ministers developed position papers in 1996 and 2002 (LANA 1996, 2002). The 2002 paper was however never published because of a lack of political consensus (Bruns, 2007). The Federal Agency for Nature Conservation also developed general standards for offsetting in the urban zone (ie under the Building Code) published in 2002 (Gerhards 2002). However, there are not currently any attempts to unify the offsetting procedure in the urban zone as defined by the 2004 Building Code.

A draft federal ordinance that defines offsetting in the rural zone was published in April 2013[85] (after several revisions), but has not been approved so far. In July 2013 the German parliament put forward a long list of requests for alterations to the draft[86], including strict limits to the power of the federal government in relation to planning applications for the electricity network, and a termination clause that automatically annuls the ordinance in 2018. The draft ordinance is no longer a political priority, and it is likely that it will not be approved in the near future. Even if it is approved, some federal states will not be obliged to adopt the national method, because federal rules mean they do not have to adapt their own state nature conservation acts.

Box 19 The proposed federal unified method for the rural zone

1. Carry out scoping impact assessment to decide whether there are expected significant impacts on species, soil, water, and climate/air, or moderate to significant impacts on landscape.

2. **Biotopes**: Map and score biotope types on a scale of 0 to 24 points according to the standard list. Adjust the biotope point scores (between 0 and 24) on the basis of survey evidence of habitat quality (structure, species composition, age etc), connectivity and location, and extent. Each biotope point score can be adjusted by 1 to 3 points up or down. Each federal state can define additional criteria by which biotope scores should be adjusted in particular areas.

4. **Species & functions**: If there are expected significant impacts on species, soil, water, and climate/air, each asset should be scored on a scale of 1 to 6 (very low, low, moderate, high, very high, significant), according to the listed criteria (functions and characteristics).

5. **Landscape**: If there are expected moderate to significant impacts on landscape aspects, a monetary compensation should take place.

For species, soil, water, climate/air and landscape, the total offset demand must be justified by qualitative reasoning. This must be additional to the biotope offset demand[87].

The proposed offset measures must be divided into planned biotope types and scored in the same way as the negative impacts, subtracting the values of the existing biotope types on the planned offset areas.

The ordinance does not specify a particular time period by which the offset biotope should achieve its expected value, but provides guidance on expected time periods for various

[85] Verordnung über die Kompensation von Eingriffen in Natur und Landschaft (Bundeskompensationsverordnung - BKompV). Entwurf 19 April 2013. http://www.bmub.bund.de/service/publikationen/downloads/details/artikel/entwurf-verordnung-ueber-die-kompensation-von-eingriffen-in-natur-und-landschaft-bundeskompensationsverordnung-bkompv-1/

[86] Hütting & Hopp September 2013 Die Bundeskompensationsverordnung – was bringt sie und wann kommt sie? ZENK Rechtsanwälte, http://www.zenk.com/fileadmin/PDF/ZENK_News/2013/September_2013/ZNL201309A06.pdf

[87] See § 7 paragraph 1 third sentence BKompV draft April 2013

biotope types. The offset area must be increased by an additional 25% for all biotopes which take more than 30 years to reach their value.

Offset measures that open up sealed soil or that remove structures that block connectivity (eg dams or weirs in rivers) are awarded a bonus of 15 points per square metre of unsealed or reconnected habitat. Guidance for calculating the area of reconnected habitat created by point measures such as wildlife tunnels is given[88].

Offset measures for significant impacts on species and functions must be carried out within the appropriate ecological range[89] and within an appropriate time period, as illustrated by examples. Exceptions must be justified for nature conservation reasons or if sufficient compensation measures are already being carried out because of legal obligations under other national laws (eg soil or air quality protection).

Reference: (BKompV 2013) draft

Typology of offset methods

The German methods can be broadly categorised by their degree of reliance on four basic procedures (Bruns, 2007; Darbi & Tausch, 2010):

- Simple compensation area coefficients / ratios for biotope types and functions (i.e. a habitat area approach according to this study's typology).
- Biotope valuation procedures, which according to this study's typology include some that are only based on the ecological value and area of biotopes, but others incorporate condition adjustments to varying degrees.
- Replacement costs approach, which estimates the cost of restoring the impacted area.
- Qualitative reasoning method (also called verbal argumentative method).

Current metrics generally use combinations of all these processes, but differ in their emphasis.

Compensation area coefficients or ratios

The Bayern method relies on a relatively simple compensation area coefficient approach. It is widely used for both rural and urban zone planning.

Box 20 The Bavarian standard metric

The impact area is divided into areas with more than 50% soil sealing (A) and areas with less than 50% soil sealing (B), and also into subareas corresponding to particular habitat types or clusters of habitat features. Each subarea is classed as either low significance (I), medium significance (II), or high significance (III) for both structure and functions. The assessor can choose the lower or upper range for each subarea depending on whether the development plan already provides mitigation measures, or whether a subarea is particularly affected by fragmentation or other specific conditions. Each affected subarea (ha) is multiplied by the compensation co-efficient and then added together to obtain the total affected area or areas that must be offset (depending on whether one or multiple like-for-like offsets are necessary). The method prescribes compensation factor ranges for each subarea derived from its soil sealing category (A) or (B) and its significance rank I, II or III. A subarea can therefore have one of six different possible compensation factor ranges (minimum 0.3x, maximum 3x or higher). A subarea should be classed with the significance allocated to the major part of its structural characteristics/functions (eg if a

[88] Annex 6 of BKompV draft April 2013

[89] In German: '*Funktionsraum*'

subarea scores high significance for landscape but medium significance for every other function and for habitat and species, its overall significance is medium). Any particular features of higher significance should then be accounted for using qualitative reasoning.

Compensation factor ranges	A – more than 50% soil sealing	B – less than 50% soil sealing
low significance (I) - intensively used arable land and species-poor grassland, canalised water courses, other biodiversity-poor agricultural or amenity landscapes, etc.	0.3-0.6	0.2-0.5
medium significance (II) - forest with non-local / non-native species, individual trees tree groups or hedges without high biodiversity interest, extensively used grassland, floodplain habitats, etc.	0.8-1.0	0.5-0.8 (0.2 for amenity and other intensively used grassland)
high significance (III) - mature semi-natural forest with a high proportion of locally appropriate tree species, mature species-rich hedgerows, copses and woodland edges, natural or near-natural freshwater landscapes, culturally significant landscapes etc.	(1.0)-3.0 (can be higher in exceptional cases)	1.0-(3.0) (can be higher in exceptional cases)

Metric and reference: Bayern (Bavaria) 'Bauen im Einklang mit Natur und Landschaft': Regelverfahren (Bavarian State Ministry of the Environment 2003), (Busse *et al* 2013)

The Bayern method was developed for ease of use by small local authorities with few qualified staff, and for its adaptability for projects in both the development zone and rural zone (Busse *et al* 2013). The metric does not rely on detailed data on the status and condition of affected habitats and species or on the proposed offset(s), but it is still necessary to gather information on species, habitats, and ecological functions to map the affected area and fulfil the legal environmental impact assessment requirements. If no regional landscape and/or species protection plan is available a more detailed site assessment is necessary.

The Bayern metric was also designed to reward high quality environmentally sensitive planning. Offsets can either move the offset area from low significance (I) to medium (II), from medium (II) to high (III), or low (I) to high (III). If the offset achieves an upgrade from I to III, the offset area does not need to be so large (Busse *et al* 2013).

There has been a broad participative debate on the use of the method in Bayern (Busse *et al* 2013). It is considered to be easily understandable by non-experts and thus the offset assessments are easily debatable in the local political process. It is argued that this transparency reinforces a broad appreciation of the cost-effectiveness of ecologically oriented planning that generates lower offset requirements (eg by reducing the sealed area through green roofs or permeable parking areas) (Busse *et al* 2013). At the same time, the simplicity and flexibility of the metric leaves a relatively large room for discretion compared to other metrics, and therefore a high responsibility on planners and local authority to demonstrate its reasonableness and proportionality. Offsets should be as close as possible in space and time to affected area, but the guidelines do not specify any method for defining offsets, though Bayern has published a list of standard costs for offset measures (Bayerisches Landesamt für Umwelt, 2011).

The Bayern method is criticised as failing to capture the links between impacts and effects because the result depends heavily on what assumptions are made to delineate and rank each subarea (Bruns, 2007). The method requires a solid qualitative textual analysis to be able to sufficiently account for specific local conditions. The size of the affected area has a big influence on the result (Bruns, 2007). Therefore large but low value impacted habitats will require a relatively high compensation in relation to their biodiversity value, whereas small-scale but severe impacts are not adequately offset. This approach does not give any guidance for what to do in the latter case. The use of higher factors for land that is already partially developed could work against the principle of discouraging sprawling development. However, the fact that the offset is derived from the total affected area plus multiplier factors means that developments on smaller areas should require smaller offsets - provided the planner selects the multipliers in such a way as to reward denser development (ie at the lower range limit). The inclusion of difficult to replace or irreplaceable habitats in the high significance class (III) was criticised as opening up the option of destroying high biodiversity value habitats if they are not protected by state laws. The counter argument is that the high offset requirement has a deterrent effect.

Habitat (biotope) valuation methods

Most of the German metrics are based on a biotope valuation method, sometimes incorporating ecosystem function metrics (see below), but differ significantly in their level of differentiation (especially regarding the number of value categories for each biotope).

An example is provided of the Sachsen method for rural zone offsetting, which uses biotope valuation based on a defined list of biotope values (adjustable to the specific case), supplemented by further valuation of ecosystem functions if functions of special or high value are impacted. The method requires that the impact area is divided into subareas with either A) impacts on assets and functions of general significance; B) impacts on assets and functions of special significance (eg priority habitats, Red List species). For the A) areas, a valuation of the impacts on biotope types is sufficient, but the use of the A) category should be justified in the impact assessment, and the method gives guidance on how to interpret whether impacts on soil, water, climate/air and landscape are significant. For the B) areas, an in-depth analysis of impacts on functions as well as biotopes is required (see Box 22).

Box 21 The Sachsen method (a): scoring impacts on biotope types

A worked example of the use of metrics to score impacts on biotope types using the Sachsen rural method is presented in Table A1.4. The example uses a fictitious development - a road crossing an agricultural landscape affecting eight different subareas and six biotope (habitat) types.

For example, the road crosses an area of the biotope type oak – hornbeam woodland, with the biotope value 27. This biotope is replaced by sealed road surface, with a value of 0, on an area of 1.35 ha, and grass verge, with a value of 5, on an area of 0.8 ha. The post-development value of this area is therefore -36.45 (total loss due to road) plus -17.6 (partial value loss due to grass) to give a net value loss of oak-hornbeam woodland of -54.05 points.

Summing the net biotope value loss of each of the eight subareas, the road will result in a total net biotope value loss of 54.95 points.

NB: In the Sachsen method, the impact on functions must also be scored (see Box 22 and Table A1.5.

References: (SMUL 2009a, 2009b), http://www.umwelt.sachsen.de/umwelt/natur/8516.htm

Table A1.4 Example of biotope valuation using the Sachsen method (part A) (SMUL 2009b)

Subarea	Negatively affected biotopes			Development			Net score	Area (ha)	Net value loss	Offset score
	Biotope Code	Biotope type	Score	Biotope Code	Biotope type	Score				
1	7 - WLE	Oak – hornbeam woodland	27	9 5 100	Road (sealed)	0	-27	1.35	-36.45	54.05
				9 5 600	Grass verge	5	-22	0.8	-17.6	
									∑54.05	
2	4 -GFY	Other damp grassland, species rich (presence of rare species such as *Orchis mascula*)	25	9 5 100	Road (sealed)	0	-25	0.3	-7.5	10.5
				9 5 600	Grass verge	5	-20	0.15	-3	
									∑10.5	
3	*03220*	Stream with straightened channel / artificial banks and semi-natural elements	18	9 5 100	Road (sealed)	0	-18	0.025	-0.45	0.58
				9 5 600	Grass verge	5	-13	0.01	-0.13	
									∑0.58	
4	4 - GFY	Other damp grassland, species rich (presence of rare species such as *Orchis mascula*)	25	9 5 100	Road (sealed)	0	-25	0.25	-6.25	8.25
				9 5 600	Grass verge	5	-20	0.1	-2	
									∑8.25	
5	*06320*	Intensively used permanent mesic grassland	10	9 5 100	Road (sealed)	0	-10	0.8	-8	9.5
				9 5 600	Grass verge	5	-5	0.3	-1.5	
									∑9.5	
6	6 5 100	Hedge (more than 60 years old)	25	9 5 100	Road (sealed)	0	-25	0.01	-0.25	0.31
				9 5 600	Grass verge	5	-20	0.003	-0.06	
									∑0.31	
7	*10120*	Intensively used arable field	5	9 5 100	Road (sealed)	0	-5	1.25	-6.25	6.25
				9 5 600	Grass verge	5	0	0.4	0	
									∑6.25	
8	*06320*	Intensively used permanent mesic grassland	10	9 5 100	Road (sealed)	0	-10	0.8	-8	10.75
				9 5 600	Grass verge	5	-5	0.55	-2.75	
									∑10.75	
										∑54.95

The Thüringen method provides an example of a biotope valuation method for urban zone planning. Impacts on functions are assessed partly through the values assigned to each biotope type, and partly through additional qualitative reasoning. Additional impacts eg on protected species must be accounted for through qualitative reasoning.

Box 22 Thüringen: biotope valuation method for the urban zone

Thüringen uses a biotope hectare method with additional qualitative evaluation of functions. Each biotope type area is scored on a basic scale with 5 levels (10, 20, 30, 40 or 50) according to the table provided. The scoring can then be differentiated on a scale between 6 and 55 according to reasoned local characteristics. Artificial or compacted soil surfaces are scored between 0 and 15; other biotope types between 15 and 55.

The value of each biotope type or subarea (=area x score) is compared to the value of the same subarea after the development has taken place (=area x score) to give a value loss for each subarea. The total value loss is the sum of all subarea losses. The offset area is scored using the biotope type values that are considered to be achievable in 30 years' time (as listed in the tables). A value gain for each subarea within the impact area and (if necessary) in external offsets is calculated and then summed to give the overall value gain for the offsets.

References: (Bruns, 2007), (Freistaat Thüringen, 2003), (Freistaat Thüringen 2005)

Use of biotope type lists with standard values

Each metric uses a different regionally developed catalogue of standard biotope types and corresponding values. Because each federal state has its own list of biotope types and values, the same or similar biotope may be valued very differently in different states, or may be defined as part of a larger category or subdivided into several types. Part of this difference may be an ecologically desirable local weighting of the importance of that biotope type in the particular area, but it is likely that some of the differences are arbitrary. These value lists have been developed by expert groups at various time periods, and using various criteria, for example:

- The Baden-Württemberg list aims to characterise the 'normal' or 'average' form of that biotope in that region when the list was compiled. It is based on the criteria naturalness, significance for threatened species, and significance as indicator of geographical and biological uniqueness (LfU Baden-Württemberg 2005b). Each of these three criteria was scored for each biotope type between 1 and 5, and the combination located on an exponential scale from 1 up to the maximum biotope score of 64. Some of the biotope types are divided into subtypes which are scored separately. The criterion regenerative capacity was not used as it was considered to be closely connected with naturalness. The process explicitly excluded cultural or historical value, as it was considered that this should be dealt with separately in the offsetting process.
- The Nordrhein-Westfalen biotope type list values biotopes according to the average of their scores for naturalness, threat status/rarity, replaceability/restoration capacity, and maturity (LANUV NRW 2008a). A few biotope types are not scored and must be assessed case-by-case. The list differs from the more generally used Nordrhein-Westfalen biotope catalogue in so far as some biotope types have been simplified whilst the woodland types have been further differentiated to account for presence of non-native species, intensity of management etc.
- The Berlin method aims to characterise the basic 'optimal' value of each biotope type using the four criteria hemeroby, which is akin to naturalness (scored from 0 to 5), presence of threatened species (animals and plants) (scored from 0 to 7), rarity or threat status of biotope

type (scored from 0 to 3), and diversity of animals and plants (scored from 0 to 5) (Land Berlin 2012). This biotope value is then differentiated according to two risk indicators on a scale from 0 to 20, and a connectivity score from 0 to 10.

The use of an exponential scale in the allocation of basic biotope values in Baden-Württemberg was considered to be more accurate because it recognises the relative ecological value of the biotopes, by ensuring that the higher scoring biotope types are progressively more valuable and therefore more demanding to offset when damaged as well as more highly ranked as offsets (LfU Baden-Württemberg 2005b).

Some of the methods offer the option to upgrade or downgrade the relative values depending on site-specific factors in order to compensate for the simplicity of the standard valuation. For example, in Nordrhein-Westfalen, each biotope type score can be adjusted by up to 2 points up or down to account for local conditions e.g. lack of naturalness due to human impacts (LANUV NRW 2008a). For woodland, arable, edge habitats and freshwater, there are defined rules for up- or down-grading. In Sachsen, biotope types are scored into 5 basic rankings for significance, and they can be further differentiated on a scale of 0 to 30 according to conservation status and other factors influencing the scale of the expected impact.

The draft federal ordinance of April 2013 provides the basis for a unified approach, attempting to combine existing methods into one coherent method for rural zone planning. The method specifies a biotope valuation procedure at the national level with a list of standard biotope types and values (on a scale from 0 to 24 including qualifiers), and also a 6-point scale for assessing the impacts on functions and landscape. The draft federal ordinance proposes a single standard biotope list for the whole of Germany. This standard list defines the biotope types on the basis of the German Red List of endangered biotopes, and assigns each a value on the basis of the sum of its biodiversity conservation value, contribution to ecological functioning, and contribution to the appreciation of nature (and therefore to some extent captures related ecosystem services)[90]. Each federal state will be required to translate their regional biotope catalogues into the national biotope catalogue, so that assessors can continue to use local biotope maps that were produced using the regional biotope catalogues.

Use of multipliers to adjust biotope valuations

Some methods incorporate multipliers to adjust metrics according to offsetting delays and risks of failure into the biotope valuation process. The Berlin method includes two explicitly defined risk indicators in the biotope valuation process, characterising restorability of the biotic components and of functions of each biotope type. Several of the guidelines (e.g. Bayern) state that such issues must be explicitly reasoned in the contextual analysis that justifies the choice of offset measures, but do not recommend the use of mathematical multipliers.

For some procedures within offset metrics, the use of multipliers is discouraged. For example, Baden-Württemberg provides a biotope valuation method for calculating the expected value of each offset measure after a 25 year period, using a list of standard biotope values. The guidance recommends that these standard biotope values are not adjusted to the particular case, or only in a minimal way, as the idea is to predict a standard expected value. Uncertainty can instead be accounted for by scoring two alternative scenarios for the development of the biotope.

Replacement costs method

Berlin uses a simple replacement costs method for small-scale low biodiversity-value impacts associated with inner city developments. Berlin has set a minimum threshold of 30.511 m^2 covered in artificial surface to trigger the offsetting requirement, and the method requires a basic mapping of biotope types, significant trees, and protected species. A biotope type map is available for Berlin. The offset measures must have at least the equivalent cost to the impacts,

[90] Defined as '*dauerhafte Sicherung der Vielfalt, der Leistungs- und Funktionsfähigkeit des Naturhaushaltes und des Erlebens und Wahrnehmens von Natur und Landschaft*' according to § 1 paragraph 1 no 1-3 BNatSchG 2009, as quoted in BKompV Begründung besonderer Teil draft April 2013

and commonly used offset measures are tree and shrub planting, green roofs, semi-permeable surfaces and other sustainable urban drainage measures. This method is regarded as providing the most flexible approach amongst German metrics[91]. It is, however, considered to be only suitable for small-scale low biodiversity-value impacts because costs are subject to a large amount of variation depending on the specific measures costed. Specific protected species issues must be considered separately (eg bat habitat protection).

Box 23 The Berlin simplified urban method (cost of restoration)

Hypothetical restoration costs are calculated for each biotic component lost (areas of biotope types and species) plus for each individual tree with a tree protection order. The calculation of costs for biotic components includes restoration/ (re)creation costs and maintenance costs (according to list of standard costs), planning costs (10% of restoration and maintenance costs). There is also an additional time lag cost (i.e. a form of multiplier) for biotope types that take more than 5 years to restore (based on European Central Bank rate of interest and according to the expected number of years - after the first five - to full restoration of that biotope type). Hypothetical restoration costs for impacts on soil, water and climate functions are calculated using the proxy of gain in artificial surfaces according to the formula €13 for each extra m^2 sealed under an artificial surface.

Reference: (Land Berlin 2012), (Busse *et al* 2013)

Qualitative reasoning (verbal-argumentative) method

The qualitative reasoning approach builds up a case-by-case argument based on expert judgement and a case-by-case selection of which natural assets, functions and biotopes, plus their possible interactions, are most affected. The Rheinland-Pfalz method relies most heavily on this approach, whilst the Brandenburg method relies on qualitative reasoning according to defined categories and standards; however, most of the German metrics guidance documents point out the importance of qualitative reasoning in accounting for impacts that cannot be quantified in the metric, such as ecological functions that do not have a spatial association.

Box 24 The Rheinland-Pfalz (1998) approach

The Rheinland-Pfalz method (1998) is the oldest currently used method in Germany, developed on the basis of consultant reports commissioned by State Agency for Environmental Protection, Health & Safety from four planning consultancies / working groups. It does not use indicator or scoring methods nor does it use a habitat value list.

The affected area is divided into subareas based on their natural asset potential (species and habitats, soil, water, air/climate, landscape), including a map of biotope types and of species that are particularly connected to priority biotope types (indicator species) and that are sensitive to disturbances. The impacts on each subarea are described through qualitative reasoning based on sensitivity and significance. Subareas are ranked as affected by complete loss, potential or partial loss of functions; insignificantly affected and not affected (on a 3 or 5 level scale). The map should indicate where subareas overlap, where functions or species interact across subareas, and the dimensions of the buffer zones. Guidance tables list examples of linkages between pressures, impacts, and interactive effects between two or more assets, to assist logical reasoning.

Reference: (LfU Rheinland-Pfalz 1998), (Lambrecht & Blischke 2005), (Bruns, 2007)

[91] Personal communication Wolfgang Wende 22 April 2014

The qualitative reasoning method is useful when data are lacking or heterogeneous or the impact is very complex (Darbi & Tausch 2011). However, it relies heavily on expert knowledge and is therefore less transparent for non-professional stakeholders, which can give an impression of arbitrariness or hide ecological inadequacies in the calculation of offset requirements.

Offsetting practice in Rheinland-Pfalz between 1997 and 2001 was critically analysed (Lambrecht & Blischke 2005). With regard to metrics, planners criticised the wide margin for interpretation and therefore the lack of transparency of the Rheinland-Pfalz method, pointing out that results based on purely qualitative reasoning are difficult to communicate clearly in particular when the required offset is large compared to the affected area. The Brandenburg guidance was evaluated after the first two years of use. Its advantage was judged to be that it takes account of unique conditions of each case and aspects that can only be assessed qualitatively. In contrast, the guidance provides detailed guidance on offset measures for animal and plant species. Its disadvantage is that it provides only a minimum level of standardisation and formal structure to provide replicability and reasonableness.

A general criticism of methods that derive functions from natural assets is that the five natural assets listed in the German law mostly contribute to the same range of environmental functions, and ecological theory does not support their compartmentalisation (Bruns, 2007). The method therefore requires a case-by-case argumentation of which functions are most significantly affected in relation to each natural asset. It is also very difficult to adequately account for all the interactions and interdependencies between functions within a reasonably short planning process.

Box 25 The Brandenburg approach

The Brandenburg approach divides each natural asset (ie species, landscape, soil, water, and air/climate) into: general significance (ie widespread biotope types with strong human influence, common species typical of nutrient-rich highly disturbed habitats); or high significance (ie threatened and/or locally typical biotope types, priority species). Each asset is also divided into: significantly affected or slightly affected (according to definitions provided for each asset). A map or plan should show areas that are particularly important for one or more assets, including protected areas, priority/typical species presence and abundance, and significant features and functions. Biotope types should be mapped according to the Brandenburg biotope type list, which categorises biotope types into 5 categories according to protected status, threat level, and restoration capacity (LUA 2011). The guidance provides examples for each category, but the quantification of impacts is not further defined. The functions associated with each asset should be selected case-by-case.

References: (LUA 2011), (MUGV 2009)

Metrics for ecosystem functions / services

The methods differ widely in their approach to metrics for ecosystem functions. Some of the methods provide explicit and detailed metrics for impacts on ecosystem functions. For example:

- In the Baden-Württemberg method (LUBW 2012), the loss of the soil's natural fertility, water cycle regulation, and pollution regulation functions are scored from 1 (minimal loss) to 5 (maximum loss) per hectare of soil lost to sealing (completely sealed soil is scored at 0). This gives a maximum function loss score of 4 points per ha or a minimum of 1 per ha. After subtraction of any mitigation and restoration measures the remaining score is weighed against the total score of an offset measure or measures, calculated in the same way. The score can also be translated into a monetary value using a standard rule of 1 to 5 Euros per m2, to give a maximum monetary value of €12 500 per ha.

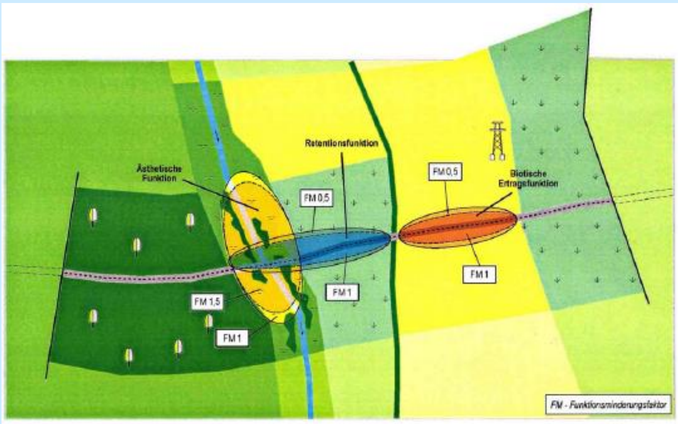
- In the Berlin method (Land Berlin 2012), ecosystem functions must always be scored using six aggregated indicators for abiotic components. The soil indicator is scored for 5 different criteria from minimum 0 (completely artificial surface) to 21. Three water indicators are scored between 0-10, 0-6, and 0-6 respectively. Two climate indicators are scored between 0-12 and between 0-10. Depending on the function, the point score for each of the abiotic component indicators is either divided by the total impact area or by a subarea. For each indicator, the scores of unaffected parts of the impact area are subtracted from these totals to give the overall net impact score for each indicator.
- In the Brandenburg method (LUA 2009), soil sealing should be compensated using a 1:1 area ratio. If this is not possible, a list of measures and corresponding ratios is provided (e.g. conversion of arable land to permanent grassland at ratio 1:0.25 for sealed soil of general significance or 1:1 for soil of high significance).

Landscape functions are usually assessed using methods that differ from the assessment of ecosystem functions.

The biotope valuation method is used as a proxy for ecosystem functions in a number of the methods (e.g. Thüringen 2005), with the specification that significant impacts on functions should be additionally assessed using qualitative reasoning. In the Sachsen method (SMUL 2009a), functions must be categorised as either: biotope-related (and therefore subsumed into the biotope indicator); non-biotope-related, i.e. not sufficiently indicated by biotope areas, and therefore requiring a separate spatial assessment; or non-spatial functions (functions that have no concrete relationship with particular areas) requiring a qualitative reasoning approach. Impacts on functions are scored using negative impact multipliers, using either 3 or 5 ranks, which can be differentiated using half ranks (0.5-5.0).

Box 26 The Sachsen method (b): scoring impacts on ecosystem functions

The use of metrics to score impacts on ecosystem functions and balance offset measures is illustrated using the Sachsen rural method in Table A1.5. This uses a fictitious development – a road crossing an agricultural landscape (see Box 17). The metric scores three functions: aesthetic function of the landscape, biotic productivity function of the soil, and the water retention /groundwater replenishment function of the affected land. The other functions are not considered to be affected significantly in the example.



ORANGE = impact on aesthetic landscape function

BLUE = impact on soil water retention /groundwater replenishment function

RED = impact on soil biotic productivity function

The aesthetic function is judged to be affected in an area where the road crosses a river, and because the effect declines with distance from the road and bridge, the impact area is divided into two. In the subarea closer to the road a function reduction factor of 1.5 is used; in the subarea farther away a function reduction factor of 1. Multiplying by the affected areas this gives a total impact of 3.65. In the fictional assessment it is argued that this impact can be offset on-site (ie in direct connection to the impact) by the removal of an electricity pylon near the road. As this is given an improvement factor of 1 on an area of 4 ha, the net gain is sufficient (ie -3.65 + 4 = +0.35).

The impacts on the soil and water functions are scored by considering the sealed area underneath the road and in close proximity with a higher impact factor of 1 and the further bordering area with a lower impact factor of 0.5. The soil productivity function is scored only for the arable field subarea, and the water retention function only for the grassland subarea bordering on the river (neither are scored for the woodland). The sum of the biotic productivity impact (1 x 1.25 ha plus 0.5 x 0.4 ha) is added to the sum of the water retention function impact (1 x 2.85 ha plus 0.5 x 1.25 ha) because the assessment judges that both cannot be completely offset directly on-site. Two on-site offset measures (afforestation and stream restoration) give a total of 2.1 points gain, which leaves a deficit of -2.83 points impact on soil and water functions to be offset elsewhere.

References: (SMUL 2009a, 2009b), http://www.umwelt.sachsen.de/umwelt/natur/8516.htm

Table A1.5 Example of metric for ecosystem function using the Sachsen method (SMUL 2009b)

impact on functions					offset measures					balance
functional subarea	Function	reduction factor	area (ha)	lost points	functional offset area	Offset	improve-ment factor	area (ha)	points gain	net points gain or deficit
FR1	aesthetic function	1	0.8	0.8	FR_A 1	removal of electricity pylon	1	4.0	4.00	**+0.35**
		1.5	1.35	2.85						
				∑3.65						
FR 2	biotic productivity function	0.5	0.4	0.2	FR_E 1	afforestation	0.5	2.7	1.35	
		1	1.25	1.25						
FR 3	water retention function	0.5	1.25	0.63	FR_E 2	stream restoration	1.5	0.5	0.75	
		1	2.85	2.85						
				∑4.93					∑2.1	**-2.83**

Combining ecosystem functions / services valuation with biotope valuation

The Berlin method for urban zone offsetting provides a standard method that combines biotope valuation with functions valuation. It assesses impacts on indicators for 19 natural assets[92] representing the biotic and abiotic components and functions plus the landscape features and services covered by the IMR.

Box 27 Berlin standard urban method: combining biotope valuation with functions valuation

The 19 aggregated indicators for natural assets are scored separately. Each indicator is scored on its own scale that combines a number of criteria at either low, medium or high.

Abiotic assets: The soil indicator is scored for 5 criteria from minimum 0 (completely artificial surface) to 21. Three water indicators are scored 0-10, 0-6, and 0-6. Two climate indicators are scored from 0-12 and 0-10. The basic scoring for the abiotic components can be done using information from the Berlin environmental maps (Umweltatlas).

Biotic assets: The assessment area is divided into biotope types according to the Berlin biotope list. A biotope type map is available for Berlin. Each biotope type is scored using four basic indicators (naturalness, presence of rare species - animals or plants, rarity or threat to biotope type, and diversity of animals and plants) plus two additional indicators that assess risk to biotopes (restorability of biotic components, restorability of functions) and a connectivity indicator. The basic indicators are scored on scales ranging from 0-3, 0-5, or 0-7. The risk indicators are scored on a scale from 0-20. Connectivity is scored from 0-10. Individual trees subject to tree protection orders are scored individually for the relevant indicators.

Landscape aspects are scored using six indicators on scales from 0-6 or 0-10.

Point scores are awarded on a per ha basis (to avoid high scores and to avoid impacts on small areas having too large an influence on the overall score). The point score for each of the abiotic components is either divided by the total impact area or by a subarea depending on the function. The total biotic point score for each biotope type is divided by the area of that biotope type, and all scores added to give a total biotic components score. The tree scores are multiplied by the stem diameter and then divided by 1000. The sum of the tree scores is added to the total biotope score. The landscape scores are divided by the total impact area. Point scores should be rounded to the nearest whole score avoiding the use of fractions. The method results in a total impact score for each of the 19 indicators.

For each of the 19 indicators, the scores of unaffected parts of the impact area (eg the undisturbed trees) are summed. For each indicator, the impact score is subtracted from the total post-impact score, and then added to give total offset requirements for the abiotic, biotic and landscape components.

The offset value can be scored in two different ways.

Using the standard method, the expected values of the offset measures are scored using similar criteria and the same scales as the impact, to give a total offset score for each of the 19 indicators. The total point score of the offset measures must equal or exceed the total point score of the impacts.

Using the simplified method, the overall impact point score can be converted into a monetary value using the following formula: sum of each point x €1.218 (rural zone) or €1.668 (urban zone) plus the value of the category of land (category I €10, category II €40,

[92] In German: Wertträgern

category III €450.

References: (Land Berlin 2012), (Busse *et al* 2013)

Keeping biotope valuation and ecosystem functions / services valuation separate

Box 28 Baden-Württemberg method: separate biotope valuation and functions valuation using ecopoints

Biotopes are classified according to the standard biotope list (including qualifiers for particular attributes or condition, and a method for assessing mixed habitat complexes). Each biotope type (and/or subtypes) has a standard value representing the average value and condition of that habitat in Baden-Württemberg. The score for each biotope is multiplied by its area. The total biotope score is the sum of all the biotope types present in the impact area. Biotopes can be scored using three different models for scoring impacts and offsets, and one for scoring offsets only (see planned biotope model for defining offsets below):

a. Standard model: quantitative biotope assessment using a 64 point scale (ie an exponential scale).

b. Detailed model: quantitative biotope assessment using 64 point scale plus multiplication factors to adjust for better or worse condition.

c. Basic model: qualitative reasoning and scoring of biotope value on 5 level scale (corresponding to the 64 point scale). For screening or for small simple plans.

The biotope score should be adapted if plants and animals on the Baden-Württemberg red list or other locally important species are present, according to a 4-point scale (extremely high significance = nationally listed species or species listed as endangered at state level; very high = state level priority species; high = priority species at regional level; medium = locally significant species).

Soil, **ground water**, and **air/climate** are scored on a 5 level scale according to specific criteria described in guidance (aggregating their functions). Surface water is generally assessed using qualitative arguments. **Landscape** is scored on a 5 level scale for each criterion including main criteria (diversity, uniqueness/history) and secondary criteria (harmony, views, naturalness, infrastructure, accessibility, scents, sounds, recreational uses). Criteria scores are combined to a joint score backed up by qualitative reasoning. A table provides short descriptions for each criterion and score.

Scores for each asset are multiplied by area to obtain overall **ecopoints** for each asset. There is no further combination of scores (ie no combination of the scores for biotope types and for functions). The scores should be displayed in a table which also indicates whether each asset is of special, medium or low significance.

The offset(s) are quantified by scoring the quality of each asset's biotopes in 25 years' time using the same method and criteria as used for the impact assessment (ie the 64-point scale). (It is assumed that all the biotope types achieve their average value within a 25 year period). The aim is to achieve offsets with high scores. The planned biotope model is a method for scoring offsets that create new biotopes (rather than restoring existing biotopes). The method scores the expected value of the biotope after 25 years on a 64 point scale. The scoring should use multipliers only in a minimal way, as the idea is to predict a standard expected value. Uncertainty can instead be accounted for by scoring two alternative scenarios for the development of the biotope. The compensation of functions is assessed separately. It should be oriented to the most significant impacts, ie if high quality soils are affected, soil offset measures are prioritised. Only in the case where this is not possible will other measures be accepted. **Four-step process to identify offset areas**: a) identify areas to offset impacts on functions (in-kind) within assessed area (ie in direct spatial connection); b) identify areas to offset impacts on functions and/or assets (in-kind) away from the assessed area (ie without direct spatial connection); c) identify areas to offset impacts on assets indirectly (out-of-kind); d) identify areas to offset impacts generally (eg for several functions together); or e) soil impacts can be (partially) offset using a monetary payment if it is not possible to offset on the site itself and if sufficient land is not available elsewhere.

References: (LfU Baden-Württemberg, 2005a, 2005b), (LUBW 2009), (LUBW 2012)

A1.4.3 Conclusions

Standardisation and adaptability requirements

A clear problem that has arisen in Germany with the implementation of offsetting has been the number and variety of metrics that have been developed. Although many are similar and relate to the biotope valuation method, the differences are sufficient to complicate offsetting procedures, especially for projects or institutions that relate to many regions and therefore use a variety of methods. This causes confusion and raises costs. For example, a comparison of the Baden-Würrtemberg method with other methods[93] using a theoretical case showed that the scoring method gives so much flexibility to expert judgement that the scoring for each biotope type can differ greatly between experts (Brauer *et al* 2006).

Some degree of standardisation is therefore needed (and proposed) but there is a risk that this could cause problems if there is not scope for adequate local adaptability. Bruns (2007) discusses the legal criteria for metrics methods in Germany. Legal certainty requires a metrics method that ensures consistency and repeatability of results even for very diverse planning situations. The method must be testable for its coherence[94]. Too much standardisation is problematic if it cannot account for local specificities; standardised methods must usually come to comparable results, but the method must include options for adapting to unusual situations.

Pros and cons of reliance on the biotope valuation method

The biotope scoring methods have the advantage of providing a standardised, easy to follow procedure that provides legal certainty. The necessary data are generally easily available in the form of habitat catalogues, maps, aerial photos and other spatial environmental data.

However, the methods can also be criticised as being variable and too simplistic. The methods depend heavily on the use of standard biotope value lists that, although generally compiled in an open expert-driven process, can include an element of arbitrariness, and also need to be regularly updated to ensure they reflect ecological realities. Ecologists may disagree on the relative value of habitats, particularly in relation to productive agricultural or forestry areas. It is in general difficult to compare the scoring for a habitat which is defined by species composition and another which is defined by structure (Bruns, 2007). As a result different biotope-based methods give different results for the same biotope type.

Another problem is that the values assigned to the biotopes are based on relative ranking, but for use as metrics are converted into an ordinal scale that may not adequately reflect the gap between the lowest ranking and highest ranking biotope types.

Biotope based methods are not suitable for assessing the full range of ecological functions. The use of biotope scoring as a proxy for ecosystem functions is based on the assumption that biotope types can adequately represent ecosystems, i.e. that they are associated with the spatial

[93] No longer in use

[94] In German: kritische Überprufung des Ergebnisses auf Schlüssigkeit vorsehen

delivery of particular ecosystem functions (Bruns, 2007). However, biotope classifications tend to rely primarily on vegetation characteristics rather than geophysical factors such as slope or soil compaction, and it is therefore questionable whether the biotope types adequately reflect the actual ecological significance of a particular affected area for all ecosystem functions, particularly hydrological functions. Some of the methods provide explicit guidance on which ecosystem functions can be subsumed into the biotope assessment, which should be assessed separately in a spatially explicit way, and which require a non-spatial assessment.

Biotope based methods are also not suitable for assessing the impacts of fragmentation, or impacts on species that are not closely connected to a particular biotope type. It is therefore generally necessary to add another layer of analysis onto the biotope valuation to account for these kinds of impacts. The reasoning justifying the results of a quantitative assessment must be plausible – it should not just be a balancing of points without sufficient explanation. Bruns (2007) proposes that the use of the metric (and environmental impact assessment report) should be understandable by educated laypeople (Bruns, 2007).

Criticism of methods for assessing ecosystem functions

A general criticism of the German methods is that they derive ecosystem functions from the five natural assets listed in the German law (soil, water, air/climate, landscape, species/habitats); however, each of these natural assets can contribute to the same range of environmental functions, and ecological theory does not support their compartmentalisation (Bruns, 2007). Ideally, each assessment should include a case-by-case argumentabout which functions are most significantly affected in relation to each natural asset. It is also very difficult to account adequately for all the interactions and interdependencies between functions within a reasonably short planning process.

The draft federal method (BKomp 2013) defines that impacts on biotope types should always be assessed, whereas impacts on animals, plants, soil, water, and climate/air only need to be assessed if significant impacts are expected[95]; landscape impacts must be assessed if a moderate to significant impact is expected. The logic behind this guidance is that it is expected that the species and environmental functions will benefit from offset measures designed to compensate for impacts on biotopes (unless there are significant impacts), whereas impacts on landscape qualities require specifically designed offset measures[96]. The German NGO NABU criticises the proposed federal method as failing to adequately account for impacts on ecological functions in areas with average or ordinary biodiversity, and as weakening the offsetting of impacts on functions, because these are ignored unless they are ranked as highly significant (NABU 2013).

Attempts to create incentives for higher value offset measures

In Nordrhein-Westfalen, offset measures that remove artificial surfaces (unseal soil) or restore connectivity (such as opening up rivers buried in culverts or creating linear features such as hedges) are assigned a double value (LANUV NRW 2008a, 2008b). The guidance also defines standard packages of offset measures for particular biotope types plus their point value, that ensure added conservation value and that upgrade the ecological value of productive land rather than taking large areas of agricultural land (for example, division of large arable fields into <1ha parcels including 3m field margins; creation of fallow with autochthonous flower rich vegetation).

[95] Defined as '*erhebliche Beeinträchtigungen besonderer Schwere*' in § 3 paragraph 3 BKompV draft April 2013; see also Zu § 8 in BKompV Begründung besonderer Teil draft April 2013

[96] See Zu § 3 in BKompV Begründung besonderer Teil draft April 2013

A1.5 The use of metrics in the Western Cape of South Africa

A1.5.1 Regulatory framework and offsetting requirements

The Western Cape of South Africa is of exceptional biodiversity importance, as it contains significant elements of two global biodiversity hotspots (the Cape Floristic Region and the Succulent Karoo) a high proportion of endemic plant species and supports a large proportion of the country's critically endangered vegetation types and threatened species (Brownlie and Botha, 2009) . In recognition of this, a number of national and provincial legal instruments aim to protect the natural environment, whilst allowing sustainable development, including the National Environmental Management Act 107 of 1998 (NEMA), and the National Environmental Management Biodiversity Act 10 of 2004 ('the Biodiversity Act'). These instruments are supported by a large number of systematic biodiversity plans and associated guidelines. Despite these laws and other initiatives to mainstream biodiversity in land use planning and impact assessment, protection of vegetation is considered to be inadequate and biodiversity considerations tend to have low priority in decision making. Thus there is an ongoing risk to priority biodiversity areas in the province (Brownlie and Botha, 2009).

However, the NEMA's principles include adherence to the mitigation hierarchy, with biodiversity offsets being seen as consistent with the need to 'remedy' significant negative impacts once efforts to avoid or minimize have been made'. The NEMA also contains the 'polluter pays' principle hence those responsible for harming the environment are required to 'pay' to remedy that harm. Furthermore, the National Biodiversity Strategy Action Plan also explicitly recognises the need for biodiversity offsets. A list of 'threatened terrestrial ecosystems' has been promulgated in terms of the Biodiversity Act, which triggers (amongst others) a requirement for Environmental Impact Assessment.

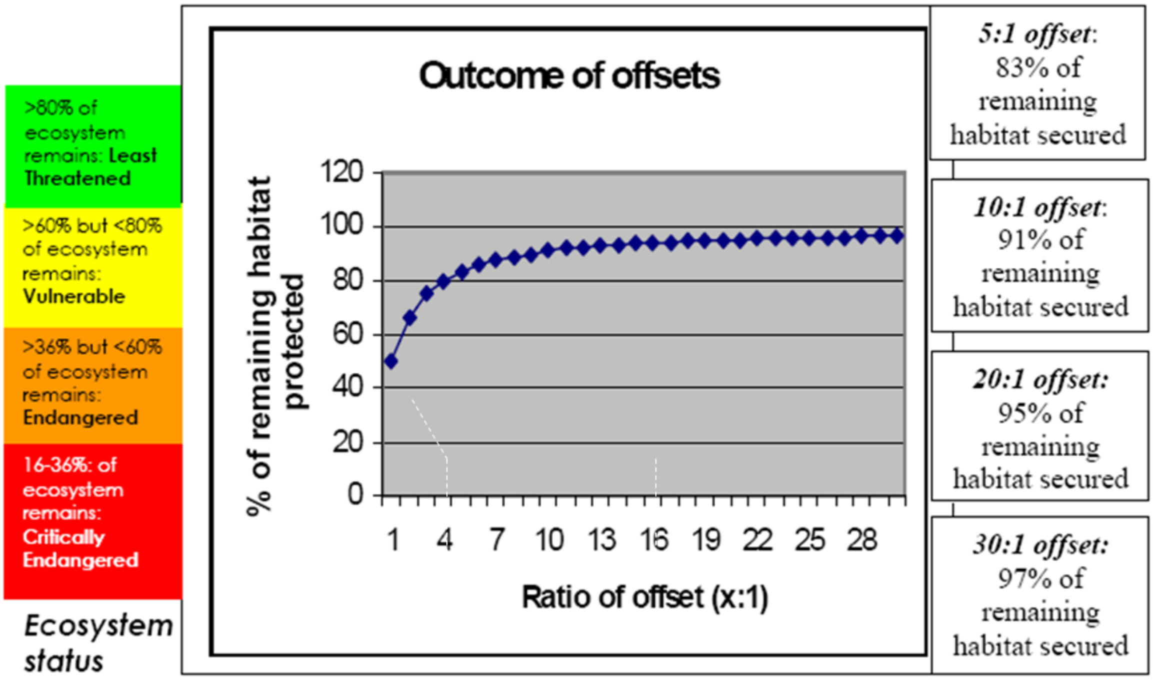
Risk-aversion offsets are primarily used in the Western Cape for all habitat other than wetlands. The rationale for this is that seeking no net loss is not realistic in such a developing country, but there is scope for improving the protection and management of many intact and threatened habitats of high conservation importance. Furthermore, restoration is virtually impossible for the majority of terrestrial ecosystems, and since the clearing of invasive alien plant species is a legal requirement for landowners such actions cannot be included in offsets because there would be no additionality. Thus the offsetting approach focuses on adding priority habitats to the conservation estate to protect and manage them in the long-term.

A1.5.2 Types of metric used and example

The metric is described below, largely drawing from Brownlie and Botha (2009) and information provided in 2014 by Susie Brownlie for this study. The principal metric used for the calculation of offset requirements is the area of the ecosystem that is affected (ie with no assessment of condition or spatial context). A 'basic ratio' for offset requirements for different levels of threatened ecosystem types is set; this multiplier takes into account the biodiversity target for the affected ecosystem and aims to ensure that no ecosystem drops below the 'endangered' category – i.e. the biodiversity target plus 15% (a safety margin). The biodiversity target takes into account the original extent of the affected ecosystem, the area already in formal protection and the % of the remaining natural area of that ecosystem that must be conserved to meet the target plus 15%. Ratios practically range from x1 to x10 depending on the level of threat. Thus, as shown in

A1.2, if 50% of the remaining area must be conserved then the ratio would be x1, if 75% of the remainder must be conserved the ratio would be x3, if 91% must be conserved the ratio would be x10).

Figure A1.2 The use of endgame multipliers in risk aversion offsets in the Western Cape province of South Africa



Source: Brownlie et al Western Cape Guidelines

This basic ratio is then adjusted on the recommendation of relevant specialists working in a specific context on a particular case according to:

- The condition of the affected habitat and potential offset site(s);
- The presence of any threatened species;
- The presence of any special habitats;
- The role of the affected area in the bigger landscape with regard to ecological processes;
- The role of the affected area in delivering ecosystem goods and/or services of socioeconomic value to local human communities and/or society as a whole;
- The probable timing of achieving the offset; and
- The level of risk or uncertainty associated with both predicting residual impacts and the probable outcome of the offset

A key feature is the ‘offset receiving area’ concept: essentially, priority areas identified in systematic biodiversity plans in the Western Cape that lie outside the protected area network are to be seen as prime offset sites by developers.

Offsets for wetlands comprise both protection (averted loss) and rehabilitation/ restoration, and their metrics are based on four main considerations:

3. Ecological functionality (supporting and regulating services)
4. Biodiversity value (ecosystem conservation value)
5. Species conservation targets
6. Provisioning or cultural services

Offset requirements for impacts on biodiversity value and species targets are calculated in a similar way to terrestrial ecosystem offsets, with the particular wetland threat status being used as the basis for an initial offset ratio. However, wetland condition and consistency with conservation plans are also taken into account. Exchange rules are used where rehabilitation/ restoration of degraded wetlands is proposed to compensate for habitat loss for species and/ or important wetlands. Thus a limit is placed on the difference in condition between offset and impact sites, in recognition that the full complement of wildlife is unlikely to return where that difference is significant.

According to Susie Brownlie (pers comm.) the methodology and metrics for determining appropriate offsets for residual impacts on species, and for provisioning/ cultural services, are relatively poorly defined at present, and left to specialists' advice.

A1.5.3 Conclusions

Although no assessment of the effectiveness of offsets and the risk aversion approach has been carried out Brownlie and Botha (2009) consider the advantages of the system to be that:

- It is relatively uncomplicated;
- It explicitly relates the size of offset to the conservation status of the impacted ecosystem;
- It sends a clear signal to developers to avoid priority biodiversity areas;
- It should significantly reduce further loss of threatened ecosystems and species;
- It introduces clear, fair (i.e. applied to all) and consistent expectations from government of developers with regard to providing biodiversity offsets and enables predictability in decision-making.

A1.6 The use of metrics in the USA

A1.6.1 Regulatory framework and offsetting requirements

Offsetting in the USA, (which is typically referred to as 'compensation') was initiated in the 1970s through the enactment of the Federal Water Pollution Control Act of 1972, commonly referred to as the Clean Water Act, which is administered by the United States Army Corps of Engineers (USACE) with oversight by the Environmental Protection Agency[97]. The Act requires the protection of certain wetlands, and Section 404 and provisions of the Food Security Act, which is administered by the Natural Resource Conservation Service (NRCS), require protection of wetlands from projects, in accordance with the mitigation hierarchy, and wetland compensation (i.e. offsets) where there are residual impacts in order to achieve no net loss of wetland area and function. Offsets are usually in the same watershed, but this is not a requirement.

Initially such offsets were carried out through an on-site or near-site project by project approach which resulted in numerous small offsets, which according to Carroll (2008) were not successful. In the 1980s, the USACE started to approve wetland mitigation banks (i.e. banked offsets), hereafter referred to as wetland banks. Federal guidance for the establishment, use and operation of mitigation banks was produced by the USACE, EPA, NRCS and US Fish and Wildlife Service (USFWS) in 1995. Permittees (i.e. developers) may create their own offsets (called permittee-responsible mitigation), or pay for offsets via third-party mitigation banks or In-Lieu Fee (ILF) programmes (i.e. in lieu of creating their own offset or buying a credit).

According to Madsen *et al* (2010), the guidance on compensatory mitigation created differing drivers and standards for the three categories of offset supply. New regulations[98] were produced in 2008 by the EPA and USACE that aimed to strengthen offsetting standards and give a stated preference hierarchy of offsets from mitigation banks (first preference) or ILF programs (second) as opposed to permittee-responsible mitigation. The rules also provide equivalent standards for all these categories of supply credits. There is also a greater watershed focus and a preference for larger, landscape-scale offsets created before the impact (versus previous guidance favouring on-site restoration).

A review of wetland banking in the US by Ecosystem Marketplace found that as of 2011 450,000 acres have been permanently protected in wetland banks in the US over the history of their use, or roughly 22,000 acres each year (Madsen *et al*, 2011). In 2010 there were 798 active banks.

Offsetting is also required in the USA to satisfy regulatory compliance for Sections 7 and 10 of the federal Endangered Species Act (ESA) of 1973, and other State and local regulations, for mitigating (i.e. offsetting) unavoidable impacts to threatened and endangered species and their habitats and other sensitive habitat areas. Offsetting of impacts on endangered species must be permitted and approved by the USFWS or National Marine Fisheries Service (NMFS) and must follow the mitigation hierarchy after which permittees may offset their residual impacts by either developing their own offset, paying into an ILF fund or buying a credit from a conservation bank[99] (i.e. an offset bank). Conservation banking is similar to wetland banking as it was based on the same concept. However, compensation for species does not have a stated 'no net loss' principle, but rather a species recovery goal, so is not strictly speaking offsetting as defined in this report.

In addition to the ESA, the California Endangered Species ACT and the California Environmental Quality Act also establish requirements for conservation banking.

[97] For key resources see http://water.epa.gov/lawsregs/guidance/wetlands/wetlandsmitigation_index.cfm

[98] Compensatory Mitigation for Losses of Aquatic Resources; Final Rule http://water.epa.gov/lawsregs/guidance/wetlands/upload/2008_04_10_wetlands_wetlands_mitigation_final_rule_4_10_08.pdf

[99] Conservation banks are also sometimes referred to as species banks, habitat banks and biodiversity banks, but here we use the term conservation banks to refer to offsetting banks under the ESA.

According to an Ecosystems Marketplace survey (Madsen *et al*, 2011), in 2011 there were 90 active conservation banks, mostly in California. These and inactive and sold-out banks include some 74,807 acres of permanently protected land in the US.

A1.6.2 Types of metric used and examples

Metrics for wetland banking

The types of metrics that are used for wetland banking in the USA are outlined below, primarily drawing on information provided by Palmer Hough (US EPA), George W. Kelly (Environmental Banc & Exchange) and Morgan Robertson (University of Wisconsin – Madison).

The assessment/metric practices in the USA vary by State, and more importantly by USACE District, of which there are 38. Some states have taken the lead in providing metrics they believe should be used (e.g. Oregon, Ohio and Florida), but in other areas the USACE decide what metrics should be used. As a result a large number of different assessment techniques have been developed for the calculation of required credits (i.e. offset gains) for wetland banks. In fact the USACE and EPA have encouraged the development of protocols that are suitable for a given region and wetland/stream type in order to avoid an undesirable one-size-fits all approach, due to the large variation in wetland/stream types and ecological properties across the USA. However, despite the variety of metrics these can be broadly divided into two types:

- Wetland ratios
- Functional assessments

The wetland ratio metric is the default method required under the mitigation regulations that should be used where a functional assessment is not possible or appropriate. It is based on an area ratio (which must be at least 1:1) of a wetland or length of streams, which is then adjusted using a variety of multipliers, which take into account:

- Wetland / stream type
- Method of compensation (wetland credit), which may be:
  - Restoration
    - Reestablishment (re-creation)
    - Rehabilitation
  - Establishment (creation)
  - Enhancement
  - Preservation
  - Buffer
- Likelihood of success, risk, difficulty
- Differences between functions lost/gained
- Temporal loss/time lag
- Distance (ie proximity to impact)

Impacts are typically initially required to mitigate at 2:1. For the mitigation credit, 50% has to come from wetland or stream restoration (to achieve no net loss) and the remainder can come from enhancement or preservation. An example of compensation type multipliers is provided below, based on the Norfolk District ratio method.

Table A1.6 Example of compensation type multipliers (based on the Norfolk District ration method)

Type of compensation	Wetland type Forested	 Scrub-Shrub	 Emergent
Restoration/Creation	2:1	1.5:1	1:1
Enhancement	-	5:1 – 9:1	-
Preservation	-	10:1-20:1+	-
Upland buffer preservation	-	15:1	-

Over half of the methods go beyond assessment of habitat suitability to encompass some assessment of wetland function, but many of these function-based methods are bounded by limitations on the type of habitat for which the method can be used and are limited in terms of the functions that are assessed, e.g. limited to avian species related functions (BBOP, 2012b). According to Morgan Robertson there has been a general transition in metrics over the last 15 years. In about 1995-2000, metrics typically had the following qualities:

- single final score (all features "rolled-up" into a final score)
- social and ecological values assessed together
- “opportunity” to provide service considered (that is, a wetland downstream of a water quality impairment might actually score higher because it has the "opportunity" to provide services)
- based on measures of structural forms rather than proxies of ecosystem function
- often translated into an area measure in the requirement of compensation
- uncalibrated to reference sites’.

An example of this would be the New Hampshire Method[100]. Although these types of metrics are still used, metrics increasingly have the following qualities:

- based on measures of ecosystem function (but still often structural proxies), or ecosystem condition, but not acreage
- often multiple final scores
- rigid separation of social and ecological values
- compared to baseline values derived from reference sites that represent realistic possible outcomes, rather than pristine states.

A typical example of this metric is the Ohio Rapid Assessment Method[101] (ORAM system), whereas, according Morgan Robertson, the Oregon Rapid Wetlands Assessment Protocol[102] (ORWAP) and Florida's Uniform Mitigation Assessment Method[103] (UMAM) are considered to be among the leading protocols that attempt to truly use functional assessments and produce a measure in units of function rather than area. The ORAM and UMAM metrics are described below.

[100] http://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/nrcs144p2_015149.pdf

[101] http://www.epa.state.oh.us/portals/35/401/oram50um_s.pdf

[102] http://www.oregon.gov/dsl/WETLAND/Pages/or_wet_prot.aspx

[103] http://www.dep.state.fl.us/water/wetlands/mitigation/umam/index.htm

The Ohio Rapid Assessment Method

The Ohio Rapid Assessment Method (ORAM) aims to enable the rapid assessment of the function and quality of wetlands in order to determine whether to permit the destruction or degradation of a wetland and to determine the appropriate level of offsetting that should be required for residual impacts (Mack, 2001). It forms part of a condition-based approach to assessing functional replacement requirements for wetland offsetting that draws on a reference data set of natural wetlands that span a gradient of human disturbance (Mack et al, 2004). The data set was used to develop the following evaluation tools were developed:

- a wetland classification scheme based on landscape position and dominant vegetation that accounts for variability in ecosystem processes (functions) and ecological services (values) of different types of natural wetlands;
- the ORAM rapid (condition based) wetland assessment tool; and
- multimetric biological indices (IBIs) and hydrological and biogeochemical indicators.

In the context of a permit application, these tools are used to assess and quantify offsetting requirements to ensure full functional replacement of the impacted wetland through the following steps:

1. The hydrogeomorphic (HGM) class and dominant plant community of the impacted wetland(s) is determined, which indicates the different ecosystem processes (functions) and ecological services (values) that each wetland type is expected to provide and must therefore be accounted for in the offsetting process.
2. The condition of the impacted wetland is assessed using the ORAM (v. 5.0) tool or a wetland IBI, which classifies wetlands according to three categories (see below) and provides a measure of "functional capacity" because it is assumed that "good" condition equates to "good" functioning.
3. The size of the impacted wetland is determined and offset ratios (set out in Ohio Administrative Code 3745-1-54[104]) are then used to quantify the area of the required offset (see below).
4. Any residual moderate to high functions or values the impacted wetland(s) may still be providing, are evaluated and taken into account, using a checklist approach with a narrative discussion (if necessary, a more detailed quantification of residual functions can be performed).
5. Requirements for offsets are specified in the permit.

The required offset must provide at least a 1:1 ratio replacement of the HGM class and dominant vegetation at the impacted wetland (i.e. of a like for like offset) of equivalent or greater “quality”. This ratio varies according to the class of impacted wetland as established by the ORAM tool (or more detailed IBI assessments for wetlands that appear to be on category boundaries), which in summary are 1) low, 2) medium and 3) high "quality." There is also a sub-category of 2, which are wetlands that are degraded but restorable. These receive the same level of regulatory protection as other Category 2 wetlands.

For Category 2 and 3 wetlands, offsets are only considered after impact avoidance and minimization measures are exhausted, (ie for unavoidable residual impacts), but this hierarchy is not as strictly applied for Category 1 wetlands as offsetting may be more appropriate (Mack, 2001). Offset ratios also vary according to whether they are on-site or off-site and whether they are through wetland restoration (the required method unless it is impractical), enhancement or preservation (ie a risk-aversion offset).

[104] http://www.epa.ohio.gov/portals/35/rules/01-54.pdf

The performance of offsets in terms of their achievement of like for like wetland functions of equivalent or greater “quality” as the impacted wetland is measured by biological, hydrological, and biogeochemical indicators derived from reference wetland data sets. These indicators then become quantitative performance standards, and mitigation monitoring is then tailored to collect the data necessary to determine if the standards have been met.

The allocation of classes through the ORAM is based on two interacting assessments: a Narrative Rating and a Quantitative Rating, as described in the ORAM user manual (Mack, 2001). The Narrative Rating is carried out first, and consists of a series of eleven questions designed to determine whether a wetland is a Category 1 (low quality) or Category 3 (high quality) wetland or to alert the assessor (referred to as ‘rater) that the wetland may be a Category 3 wetland. The first four questions ask the assessor to consult the U.S. Fish and Wildlife Service, the State of Ohio’s Natural Heritage Database and/or other readily available information sources to determine whether the wetland in question has the characteristics of a Category 3 wetland. The remaining questions focus more on whether the wetland in question is of very poor quality, locally scarce or scarce throughout Ohio and also allows for the identification of particular types of wetlands which often have high levels of diversity, high native species richness, or high functional values. For example, Question 6 asks “Is the wetland a peat-accumulating wetland that has 1) no significant inflows or outflows, 2) >30% cover of acidophilic mosses, particularly *Sphagnum* spp., 3) at least one of species listed in Table 1 in the wetland, and 4) <25% cover of the invasive species listed in Table 1?”

The Quantitative Rating is a combined metric consisting of six-components as set out in Table A1.7 (with scoring sheets provided further below). These scores are simply summed, and have a potential score of 100. Guidance on the assessment of each metric component is provided in the ORAM user manual. It should be noted that the assessment does not attempt to directly assess the importance of the wetlands functions, but uses indicators that aim to ensure that wetlands that have moderate to high quality functions and habitats will be rated as Category 2 or 3 wetlands, while highly degraded systems with minimal functions or habitats will be rated as Category 1 wetlands.

Table A1.7 ORAM Quantitative Assessment metric scores

Adapted from (Mack, 2001)

Metric component	Title	Sub-component	Sub-component maximum score	Component maximum
1	Wetland size	None	6	6
2	Upland buffers and surrounding land use	2a. Average buffer width	7	14
		2b. Surrounding land use	7	
3	Hydrology	3a. Sources of water	7	30
		3b. Connectivity	11	
		3c. Maximum water depth	3	
		3d. Duration inundation or saturation	3	
		3e. Modifications to natural hydraulic regimes	4	
4	Habitat alteration and development	4a. Substrate disturbance	4	20
		4b. Habitat development	7	
		4c. Habitat alteration	9	
5	Special wetland communities	None	10+/10-	10
6	Vegetation, interspersion & microtopography	6a. Wetland vegetation communities	18	20
		6b. Horizontal community interspersion	5	
		6c. Presence of table 1 invasive	-5	
		6d. Microtopography	12	

The ORAM Quantitative Rating indicates the wetland category as below[105]

Category	ORAM v5.0 Score
1	0 – 22.9
1 or 2 gray zone	30 – 34.9
modified 2	35 – 44.9
2	45 – 59.9
2 or 3	60 – 64.9
3	65 – 100

There are three possible results and follow up actions from the Narrative Rating:

- The wetland is a Category 1 wetland; in which case it is considered to be a Category 1, unless the wetland scores above the Category 1 threshold on the Quantitative Rating. In that case the assessor should re-evaluate the category of the wetland using the narrative criteria in OAC Rule 3745-1-54(C) and further evaluate the wetland using detailed assessments, including determining a wetland IBI score for that type of wetland.
- The wetland should be evaluated for possible Category 3 status; in which case the assessor should 1) evaluate the category of the wetland using the narrative criteria in OAC Rule 3745-1-54(C) and 2) evaluate the category of the wetland using the Quantitative Rating. If the wetland is determined to be a Category 3 wetland using either of these, it is a Category 3 wetland.
- The wetland is a Category 3 wetland. In this situation, the wetland should be considered a Category 3 wetland unless the wetland scores in the Category 1 range on the Quantitative Rating. In that case the assessor should re-evaluate the category of the wetland using the narrative criteria in OAC Rule 3745-1-54(C) and further evaluate the wetland using detailed biological or functional assessments, including determining the IBI score for that type of wetland.

Thus the ORAM scoring method aims to ensure consistency in assessments and to gather further investigations using more complex and reliable assessments methods if there is a risk of an incorrect categorisation. In addition, to ensure a robust categorization, or one that is in accordance with the precautionary principle, a pre-determined set of intermediate scores, are referred to as the "gray zone". If an ORAM assessment falls within the gray zone, the assessor can do either of the following:

- Assign the wetland to the higher of the two categories, e.g. if the wetland is in the gray zone between Category 1 and 2, the assessor would assign the wetland to Category 2;
- Assess the quality of the wetland using a non-rapid method (ie not the ORAM), that provides a detailed functional and/or biological assessment of the wetland and use this information in conjunction with any wetland indices of biotic integrity, the narrative criteria to assign the wetland to a category.

[105] Based on the thresholds given in Ohio Environmental Protection Agency website http://www.epa.ohio.gov/dsw/401/ecology.aspx#149364493-ohio-rapid-assessment-method-oram

By definition the ORAM is a rapid assessment system, and therefore is simple and quick to carry out and likely to be of low cost. The required time to assess wetlands will vary according to their circumstances, such that the Narrative and Quantitative Ratings may be answered in a few minutes for small and isolated wetlands, but take will take several hours for large wetlands, or wetlands that are part of a complex of wetlands that must be scored together (Mack, 2001). However, the assessment may trigger requirements for further in-depth surveys and evaluations if the wetland category is unclear or if it is high functional value.

ORAM v. 5.0 Field Form Quantitative Rating

Site:	Rater(s):	Date:

Metric 1. Wetland Area (size).

max 6 pts. subtotal

Select one size class and assign score.

- ☐ >50 acres (>20.2ha) (6 pts)
- ☐ 25 to <50 acres (10.1 to <20.2ha) (5 pts)
- ☐ 10 to <25 acres (4 to <10.1ha) (4 pts)
- ☐ 3 to <10 acres (1.2 to <4ha) (3 pts)
- ☐ 0.3 to <3 acres (0.12 to <1.2ha) (2pts)
- ☐ 0.1 to <0.3 acres (0.04 to <0.12ha) (1 pt)
- ☐ <0.1 acres (0.04ha) (0 pts)

Metric 2. Upland buffers and surrounding land use.

max 14 pts. subtotal

2a. Calculate average buffer width. Select only one and assign score. Do not double check.

- ☐ WIDE. Buffers average 50m (164ft) or more around wetland perimeter (7)
- ☐ MEDIUM. Buffers average 25m to <50m (82 to <164ft) around wetland perimeter (4)
- ☐ NARROW. Buffers average 10m to <25m (32ft to <82ft) around wetland perimeter (1)
- ☐ VERY NARROW. Buffers average <10m (<32ft) around wetland perimeter (0)

2b. Intensity of surrounding land use. Select one or double check and average.

- ☐ VERY LOW. 2nd growth or older forest, prairie, savannah, wildlife area, etc. (7)
- ☐ LOW. Old field (>10 years), shrub land, young second growth forest. (5)
- ☐ MODERATELY HIGH. Residential, fenced pasture, park, conservation tillage, new fallow field. (3)
- ☐ HIGH. Urban, industrial, open pasture, row cropping, mining, construction. (1)

Metric 3. Hydrology.

max 30 pts. subtotal

3a. Sources of Water. Score all that apply.

- ☐ High pH groundwater (5)
- ☐ Other groundwater (3)
- ☐ Precipitation (1)
- ☐ Seasonal/Intermittent surface water (3)
- ☐ Perennial surface water (lake or stream) (5)

3b. Connectivity. Score all that apply.

- ☐ 100 year floodplain (1)
- ☐ Between stream/lake and other human use (1)
- ☐ Part of wetland/upland (e.g. forest), complex (1)
- ☐ Part of riparian or upland corridor (1)

3c. Maximum water depth. Select only one and assign score.

- ☐ >0.7 (27.6in) (3)
- ☐ 0.4 to 0.7m (15.7 to 27.6in) (2)
- ☐ <0.4m (<15.7in) (1)

3d. Duration inundation/saturation. Score one or dbl check.

- ☐ Semi- to permanently inundated/saturated (4)
- ☐ Regularly inundated/saturated (3)
- ☐ Seasonally inundated (2)
- ☐ Seasonally saturated in upper 30cm (12in) (1)

3e. Modifications to natural hydrologic regime. Score one or double check and average.

- ☐ None or none apparent (12)
- ☐ Recovered (7)
- ☐ Recovering (3)
- ☐ Recent or no recovery (1)

Check all disturbances observed

☐ ditch	☐ point source (nonstormwater)
☐ tile	☐ filling/grading
☐ dike	☐ road bed/RR track
☐ weir	☐ dredging
☐ stormwater input	☐ other_________________

Metric 4. Habitat Alteration and Development.

max 20 pts. subtotal

4a. Substrate disturbance. Score one or double check and average.

- ☐ None or none apparent (4)
- ☐ Recovered (3)
- ☐ Recovering (2)
- ☐ Recent or no recovery (1)

4b. Habitat development. Select only one and assign score.

- ☐ Excellent (7)
- ☐ Very good (6)
- ☐ Good (5)
- ☐ Moderately good (4)
- ☐ Fair (3)
- ☐ Poor to fair (2)
- ☐ Poor (1)

4c. Habitat alteration. Score one or double check and average.

- ☐ None or none apparent (9)
- ☐ Recovered (6)
- ☐ Recovering (3)
- ☐ Recent or no recovery (1)

Check all disturbances observed

☐ mowing	☐ shrub/sapling removal
☐ grazing	☐ herbaceous/aquatic bed removal
☐ clearcutting	☐ sedimentation
☐ selective cutting	☐ dredging
☐ woody debris removal	☐ farming
☐ toxic pollutants	☐ nutrient enrichment

subtotal this page

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ORAM v. 5.0 Field Form Quantitative Rating

Site:	Rater(s):	Date:

subtotal first page

max 10 pts. subtotal

Metric 5. Special Wetlands.

Check all that apply and score as indicated.

- Bog (10)
- Fen (10)
- Old growth forest (10)
- Mature forested wetland (5)
- Lake Erie coastal/tributary wetland-unrestricted hydrology (10)
- Lake Erie coastal/tributary wetland-restricted hydrology (5)
- Lake Plain Sand Prairies (Oak Openings) (10)
- Relict Wet Prairies (10)
- Known occurrence state/federal threatened or endangered species (10)
- Significant migratory songbird/water fowl habitat or usage (10)
- Category 1 Wetland. See Question 1 Qualitative Rating (-10)

max 20 pts. subtotal

Metric 6. Plant communities, interspersion, microtopography.

6a. Wetland Vegetation Communities.
Score all present using 0 to 3 scale.

- Aquatic bed
- Emergent
- Shrub
- Forest
- Mudflats
- Open water
- Other______________

6b. horizontal (plan view) Interspersion.
Select only one.

- High (5)
- Moderately high(4)
- Moderate (3)
- Moderately low (2)
- Low (1)
- None (0)

6c. Coverage of invasive plants. Refer to Table 1 ORAM long form for list. Add or deduct points for coverage

- Extensive >75% cover (-5)
- Moderate 25-75% cover (-3)
- Sparse 5-25% cover (-1)
- Nearly absent <5% cover (0)
- Absent (1)

6d. Microtopography.
Score all present using 0 to 3 scale.

- Vegetated hummucks/tussucks
- Coarse woody debris >15cm (6in)
- Standing dead >25cm (10in) dbh
- Amphibian breeding pools

Vegetation Community Cover Scale

0	Absent or comprises <0.1ha (0.2471 acres) contiguous area
1	Present and either comprises small part of wetland's vegetation and is of moderate quality, or comprises a significant part but is of low quality
2	Present and either comprises significant part of wetland's vegetation and is of moderate quality or comprises a small part and is of high quality
3	Present and comprises significant part, or more, of wetland's vegetation and is of high quality

Narrative Description of Vegetation Quality

low	Low spp diversity and/or predominance of nonnative or disturbance tolerant native species
mod	Native spp are dominant component of the vegetation, although nonnative and/or disturbance tolerant native spp can also be present, and species diversity moderate to moderately high, but generally w/o presence of rare threatened or endangered spp
high	A predominance of native species, with nonnative spp and/or disturbance tolerant native spp absent or virtually absent, and high spp diversity and often, but not always, the presence of rare, threatened, or endangered spp

Mudflat and Open Water Class Quality

0	Absent <0.1ha (0.247 acres)
1	Low 0.1 to <1ha (0.247 to 2.47 acres)
2	Moderate 1 to <4ha (2.47 to 9.88 acres)
3	High 4ha (9.88 acres) or more

Microtopography Cover Scale

0	Absent
1	Present very small amounts or if more common of marginal quality
2	Present in moderate amounts, but not of highest quality or in small amounts of highest quality
3	Present in moderate or greater amounts and of highest quality

End of Quantitative Rating. Complete Categorization Worksheets.

The Florida Uniform Mitigation Assessment Method

The Florida metric utilises a Wetland Rapid Assessment Procedure (WRAP) which uses a combination of desk and field-based assessments to arrive at functional capacity score for the impacted wetland pre- and post-development and the compensation (offset) wetland pre- and post-offsetting. This leads to a WRAP rating index, which establishes a numerical ranking for individual ecological and anthropogenic factors that can strongly influence offsetting success, which include:

- wildlife utilization
- overstory/shrub canopy
- vegetative groundcover
- adjacent upland support/buffer
- hydrology
- water quality input/Treatment systems

The metrics also take into account a Site Suitability Index, which aims to incentivise the strategic location of offsets (by giving extra credits to sites in desired locations or with desired spatial attributes), and includes a risk multiplier.

However, it should be noted that the ORWAP and UMAM methods are designed for regulatory purposes, and are not therefore likely to reflect the state-of-the-art in the scientific assessment of wetland functions, often for very good reasons. Indeed, practicality is clearly a key driver of the design in offsets, as for example, in Oregon a rule of thumb is that any useful assessment system should follow the 2/2/2 rule, that is, takes no longer than 2 hours to assess a site, by a team of 2 people with no more than 2 days of training. Although it is not clear how this is affected by wetland size, it suggests that some wetland functions may overlooked and therefore not adequately compensated for.

More examples of wetland and stream assessment approaches used in different states can be seen on the RIBITS database[106] (by searching by State and clicking on the Assessment Tools tab).

Species-based metrics: Habitat Evaluation Procedures

The most sophisticated metrics have been developed for species offsetting, and include the widely used Habitat Evaluation Procedures (HEP). The HEP was initially developed by the US Fish and Wildlife Service (USFWS) in 1976[107]. It has been widely used by land managers in the USA and adapted for use in other countries such as the UK (see section A1.2.2), as it allows the impacts of a wide variety of land use changes to be assessed in a standardised way.

The procedure is described in section 3 on species metrics, but in summary the rationale for the HEP is that areas affected by projects and those identified for offset activities can contain various habitats, and these habitats can have different suitabilities for species that may occur in that area that can be quantified through habitat suitability models, resulting in a Habitat Suitability Index (HSI) that ranges from 0 to 1. Provided that the extent of the different habitats can be measured, the overall suitability of an area for a species can be represented as a product of the area of each habitat and the HSI index for each habitat for the species, which is referred to as Habitat Units (HUs).

[106] http://geo.usace.army.mil/ribits/index.html

[107] http://www.fws.gov/policy/esm102.pdf

The early HEP analysed habitat types, but now focuses on selected evaluation species and uses habitat modeling to do this. Support for carrying out HEPs is now available, including detailed accounting spreadsheets, manuals and training programmes[108].

[108] Available at http://www.fort.usgs.gov/Products/Software/HEP/