An analysis of the empirical evidence on markets for biodiversity offsets

# Rationale of the review

Biodiversity offsetting aims to reconcile the mounting pressures for urban development and agricultural expansion with the need to restore biodiversity (Ten-Kate *et al.* 2010). Offsetting is considered the final step in the mitigation hierarchy once all other steps (avoid, minimize, restore) have been undertaken by the proposed new development and provide measurable conservation gains to compensate for significant, residual impacts on biodiversity due to new development activities (BBOP 2009; Alridge *et al.* 2019). The majority of offset policies historically have targeted no net loss of biodiversity, where losses due to development are matched by gains in biodiversity elsewhere (zu Ermgassen *et al.* 2019). More recently, the focus in academia and policy has been shifting towards Net Positive Impact and biodiversity net gain. Net gain requires actions that ensure recreated or restored habitats exceed those lost in terms of potential biodiversity outcomes (that is, gains outweighing losses in some agreed metric) (CIEEM, CIRIA and IEMA, 2016; Bull and Brownlie 2017; Moilanen and Kotiaho 2020)

There have been numerous reviews of biodiversity offsetting, predominantly from the ecological perspective concerning the effectiveness of restoration, the functionality of restored systems compared to natural systems, how to assess ecological equivalency, and additionality (Bull *et al.* 2013; Bull *et al.* 2015; Bull *et al.* 2017; Maron *et al.* 2012; Quétier *et al.* 2014; Maron *et al.* 2015; Maron *et al.* 2016). Perspectives have also been offered from environmental law and policy (Salzman and Ruhl 2000; Salzman and Ruhl 2006). However, there have been few contributions from the economics literature on the subject (Needham *et al.* 2019).

In this review, we focus on markets for biodiversity offsets. This market-based instrument is created when multiple buyers and sellers of offsets interact with others through a trading process, typically moderated by an offset bank or regulator (Simpson *et al.* 2021). Landowners can choose to manage land for conservation, generating offset credits which can then be sold to a developer who is required to mitigate development impacts, for example from house building, on some measure of biodiversity. By establishing an appropriate rate of exchange between sellers and buyers, markets can, in theory, achieve NNL or a NG in biodiversity within some defined area at least cost. Through the creation of economic incentives for conservation, incentive mechanisms such as markets for offsets encourage private landowners and firms to take costly actions that benefit biodiversity (Kangas and Ollikainen 2019).

Within offset policies, we focus in particular on the evidence base for how technical choices (such as biodiversity metrics, proximity constraints and treatment of ecological equivalence) vie with practical considerations (such as the expected duration of contracts, policy stability, and presence of localised expertise) to affect the outcomes of offset markets and trades. It is known that a perennial problem in the emergence of nascent offset markets has been a shortage in the supply of viable offset receptor sites. We consider whether key barriers to incentivising land managers into offsetting lean more towards the theoretical or the practical. Substantial research has been carried out into important design parameters for biodiversity offset programmes, focussing on the theoretical (Bull *et al.* 2013) to those focussed around implementation (Moilanen and Kotiaho 2018; White *et al*. 2021). But offsetting requires land, and there are some important disincentives for land managers to participate: particularly the lack of flexibility in the measures required on offset sites to meet a no net loss or net gain objective; the longevity often specified by offset policy, which might potentially require offset management activities to be locked in for many decades; and the lack of stability these policies can have (e.g. Damiens *et al*. 2020). There is a temptation to relax policy rules in order to facilitate greater offset site availability, but this runs the risk of undermining the physical validity of the policies themselves (zu Ermgassen *et al*. 2020). Our review seeks to collate and improve understanding of the incentives and disincentives for land managers to participate in offset markets and explore potential solutions for increasing participation without jeopardising the fundamental objectives of the policy.

Therefore, our study provides an up-to-date overview of existing literature on the empirical implementation of markets for biodiversity offsets.

Our objectives are:

1. to provide a general overview of the design characteristics of biodiversity offset markets;
2. to identify the barriers to landowners for creating biodiversity offsets;
3. to identify factors that drive participants to buy and sell biodiversity offset credits, and
4. identify existing research gaps.

To fulfil these objectives, we found a scoping review approach to be the most relevant method. A scoping review is a systematic literature review approach that seeks to represent, evaluate, and describe the contents of various previous studies to understand the evidence presented while identifying potential knowledge gaps (Arksey and O’Malley 2005).

## Research questions:

Besides providing a detailed account of evidence within each study (see Supplementary Material 1 – Excel sheet), we sought to answer the following specific questions:

1. How do technical choices (such as biodiversity metrics, proximity constraints and treatment of ecological equivalence) vie with practical considerations (such as the expected duration of contracts, policy stability, and presence of localised expertise) to affect the outcomes of offset markets/trades?
2. Understand how incentives and disincentives affect the decisions of land managers to participate in offset markets, and explore potential solutions for increasing participation without jeopardising the fundamental objectives of the policy.

# Literature search strategy development

Our search strategy started with defining the individual terms associated with biodiversity offsetting and biodiversity offset markets (Table 1). We also identified that biodiversity offsetting can encompass a wide range of mechanisms including compensatory mitigation, in-kind compensation, mitigation banking, habitat banking, species banking, and wetland banking (Lapeyre *et al.* 2015). This is particularly evident when reviewing the early evidence of biodiversity offset markets in the USA (Wetland Mitigation Banking) and offset markets in Australia (BioBanking). In our review, we focus specifically on mitigation banking, wetland banking and biodiversity offset markets.

**Table 1: Definition of key terms in payment for results agri-environment schemes context**

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| **Term** | **Definition** | **Source (s)** |
| Biodiversity offsetting | Biodiversity offsets provide measurable conservation gains to compensate for significant, residual impacts on biodiversity due to new development activities. | BBOP (2009) |
| Market-based instrument | Market-based instruments seek to address the market failure of 'environmental externalities' either by incorporating the external cost of production or consumption activities through taxes or charges on processes or products, or by creating property rights and facilitating the establishment of a proxy market for the use of environmental services. | OECD (2007) |
| Biodiversity offset markets | Biodiversity offset markets are created when multiple buyers and sellers of offset credits interact with others through a trading process, typically moderated by an offset bank or regulator | Needham *et al.* (2019) |
| No net loss | NNL policies are based on the principle that biodiversity is as a minimum left no worse off after development than before. | zu Ermgassen *et al.* (2019) |
| Net gain | To achieve Net Gain, the biodiversity status quo must be improved, either by overcompensating for loss in the biodiversity values affected or by ensuring no net loss in those values and then providing additional gains in other biodiversity values. | Bull and Brownlie (2017) |

## Database search process

We searched our literature from two main databases, Scopus and Web of Science. Scopus database holds papers of satisfactory quality in diverse specialities: natural sciences and social sciences while Web of Science is the most reliable worldwide citation database. We limited our search to English papers with no particular restriction to geographical scope and publication date. During the database search process, we use Boolean “OR” and “AND” to combine the different keywords that we developed from the literature. In the event keywords had multiple endings e.g. offse**t** or offset**s,** we used the asterisk (\*) sign to shorten such words. To limit our search to the keywords provided, we used the double quotation marks “…”. Table 2 shows a summary of our search word combinations.

**Table 2: Keywords and search strings**

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| --- |
| biodiversity AND offset\* AND market\*  wetland\* AND mitigation AND bank\*  habitat\* AND offset\* and bank\*  biodiversity AND offset\* AND bank\* |

## Data screening and inclusion criteria

All relevant articles were transferred from the databases to online endnote software. The first step was to remove duplicates. We further screened relevant articles based on titles and abstracts followed by an in-depth review of full texts. The studies that we selected for review had to meet the following criteria. First, they had to be from a peer-reviewed journal and in the English language. Second, they had to have findings achieved from a qualitative or quantitative analysis of secondary or primary data on biodiversity offset markets. We excluded systematic reviews, book chapters, reports, editorials, commentaries, and studies not focusing on the empirical application of biodiversity offset markets.

## Data extraction process

Our data extraction process was comprehensive and was made up of three key parts. The first part contained data on the study characteristics fields such as the name of the first author, year of publication, title, the objective of the study, study type, study area, data type etc. The second part contained detail on offset design parameters and whether these were analysed as part of the research paper (metric, multiplier, time period, scale, contract design, follow up monitoring and enforcement). The third part reports the main conclusions of the study, alongside the strengths and weaknesses. We provide a detailed summary of these data in an Excel sheet (see supplementary material 1).

## Data analysis strategy

We thematically and descriptively synthesized our literature to draw interpretations and conclusions.

# Results

Our search using different combinations of our keywords in Web of Science and Scopus gave us a total of 608 articles out of which 42 articles met our set criteria for inclusion in our review (Figure 1). Studies selected for this review were published between 1999 – 2022 and were carried out in nine countries. 20 of the studies identified were from the USA dating back to 1999, which is unsurprising given the history of wetland mitigation banking in the country. 8 focussed on the UK, with the first study published in 2014, reflecting the shift in favour of biodiversity offsetting from 2011 within UK environmental policy. Other countries studied included Australia, France, Germany, Canada and Sweden.

Figure 1: The literature screening process

We grouped our articles by broad themes related to the design and implementation of markets for biodiversity offsets based on key design parameters previously identified in biodiversity offset market literature (Needham *et al.* 2019). These themes covered the technical choices (metrics, proximity and ecological constraints) and practical considerations (contracts, policy stability and expertise). Given the prominence of articles from the USA and the UK, we first outline policies relating to offset markets in these countries, before discussing literature related to each of the specific themes.

**3.1 Biodiversity offset markets in the USA**

The USA adopted Wetland Mitigation Banking (WMB) following Section 404 of the Clean Water Act (1977) which required ‘no net loss of wetland acreage and function'. The first WMB banks were established in the 1980s (Hough and Robertson 2009). WMB credits were initially based on habitat acreage, but more recent approaches have sought to consider wetland functioning, accounting for the effects of offsetting actions on water quality, flood storage and flow reductions and habitat quality (Doyle and Shields 2012). Multipliers (also referred to as trading ratios in the literature) are applied to account for differences in ecological criteria between the developed site and the offset bank. Typically, creation or re-establishment has the lowest requirements (1:1) whilst banks which preserve existing sites have the highest trading ratios (3:1) (Environmental Protection Agency 2022). Once a credit is sold the area should typically be protected in perpetuity through conservation easements.

The first banks established throughout the 1980s were publicly sponsored single-user banks with state agencies and large organisations stockpiling credits for their own later use. From 1991 onwards, entrepreneurial banks were established and private offset providers could sell credits to any developer. As of May 2022, there were over 2387 banks and bank sites loaded into the RIBITS (Regulatory In-lieu fee and Bank Information Tracking System)[[1]](#footnote-1) database. Of these 1864 are private/entrepreneurial banks. We explore the lessons learned from the USA WMB markets in the following sections.

## 3.2 Biodiversity Offset Markets in the UK

In the UK the idea of a biodiversity offsetting system was announced in the Government’s Natural Environment White Paper in 2011 (Defra 2011). Prior to this, there had been a growing movement through the Countryside and Rights of Way Act (2000), the Natural Environment and Rural Communities Act (2006) and associated planning policy for some planning authorities to seek ecological compensation for impacts on a broader spectrum of biodiversity than before, and more specifically, to explore options for offsetting (Treweek *et al*. 2009). It was felt that biodiversity offsetting could overcome some of the issues associated with Section 106 agreements. These legally binding agreements between the developer and local planning authority offer management or financial support to offset adverse environmental effects. However, in some circumstances it was felt that Local Authorities spent a long time detailing such agreements but often failed to enforce these once planning permission was granted, leaving a substantial gap between what was promised by developers and what was delivered (Briggs *et al*. 2009).

In April 2012 the UK government established pilot biodiversity offset markets for six local authorities that ran for two years with the overall aim of the biodiversity offsetting pilots was “to develop a body of information and evidence to inform a future decision about whether to use biodiversity offsetting across England (Defra 2011). Within the pilot areas, developers who were required to provide compensation for biodiversity loss could choose to do so through offsetting. The pilot targetted NNL of biodiversity, with the aim that offsets should be additional i.e. it should expand and restore habitats and enhance the ecological network by joining linking conservation areas. The aim of the scheme was to be managed at the local level and not create additional administrative burdens for developers and local authorities (Defra 2012).

The offset metric was based on habitats and took into account species richness, diversity, rarity (at local, regional, national and international scales) and the degree to which the habitat supported species rarely found in other habitats. Site condition was assessed using the same tool as the Higher Level agri-environment Scheme. Multipliers were applied to take into account the risk associated with restoring or expanding certain habitats and to take into account locational differences between the offset site and the original site. There was an expectation that multipliers associated with time differences between the habitat destruction and offset provision would incentivize the creation of habitat banks by starting the offset development well ahead of the proposed project (Defra 2012b).

Following the completion of the pilots a variety of technical reports (Baker *et al.* 2016; Defra 2016) and articles (Lockhart 2015; Sullivan and Hannis 2015) have been published reviewing various aspects of the scheme: most notably the offset metric and market set up (which we discuss in later sections of this paper).

Biodiversity offsetting is now embedded within the UK policy framework through the Environment Act 2021. Developments under the Town and Country Planning Act (i.e., nearly all residential, commercial, and mining construction) in England are now required to deliver a mandatory net gain in biodiversity (zu Ermgassen *et al*. 2020). The 25 Year Environment Plan specifically identifies that this net gain could be delivered by “habitat banks, land-owners or brokers” as part of a flexible market (HM Government 2018).

## Offset Implementation: emerging lessons

Of the 42 papers selected for this review, 23 took used a case study analysis approach to understand and analyse multiple aspects of biodiversity offset markets. These papers focussed on practical considerations of offset markets and aimed to evaluate the success and failure of existing biodiversity offset schemes and apply lessons learned to the emerging policy frameworks. One of the main methods used was the development of databases of existing offset or compensatory mitigation schemes, at either the country level (e.g. biodiversity offset policy in England as undertaken by zu Ermgassen *et al.* 2021) or at the state level (for wetland mitigation banking in the USA as undertaken by BenDor & Doyle 2009). Through the development of databases of existing offset schemes and compensatory mitigation, the authors were able to assess the extent that which the existing scheme design was able to deliver biodiversity offsets and what the main factors were driving this. Despite the differences in terms of the country, policy design and era similar findings were evident in the works of BenDor & Doyle (2009) and zu Ermgassen *et al.* (2021): a key barrier to implementing offsets at the practical level is the lack of regulatory capacity to implement the programme. In particular, the follow-up monitoring and enforcement of newly created offset sites. There is a lack of connectivity between the relevant institutions, stakeholders and regulatory partners. There is also a lack of consideration for future land uses and this should be a required component of restoration site plans and the broader region.

A second method used to analyse existing and future policy directions for offsetting were semi-structured interviews, workshops and stakeholder surveys with relevant parties (for example White *et al.* 2021; Blicharska *et al.* 2022; Taherzadeh and Howley 2018; Grimm 2020). Focussing on the USA initially, multiple surveys were undertaken with mitigation bankers between 2005 and 2010 that focussed on ecological and financial risks associated with designing a new wetland mitigation banks (Robertson 2009; BenDor and Riggsbee 2011) with a particular emphasis on the size and scale of the market and the effects of the implementation of the 2008 ruling to establish what was essentially a mandatory market for wetland mitigation banked credits ahead of developer responsible mitigation. We discuss this in detail within section XX. Our literature search identified a second wave of surveys that were undertaken between 2015 and 2021with by multiple stakeholders involved in species conservation banking (White *et al.* 2021; Grimm 2020; Leveral *et al.* 2017; Vaissière *et al.* 2017). Stakeholders view successful species conservation policies as those that were based on landscape-scale mitigation goals and used conservation easements to protect the land in perpetuity. As evident in the database analysis approach, concerns focussed on the lack of consistent evaluation across bank sites, with a lack of regulatory follow-up monitoring and enforcement following the bank’s approval. This was especially evident in cases where a large number of credits were released upfront when the bank agreement was signed but before restoration actions were undertaken.

For England, we identified a series of practitioner surveys related to the initial design of biodiversity offsetting pilot studies (Lockhart 2015; Sullivan and Hannis 2015; Carver and Sullivan 2017) and one considering the future of offsets in England (Taherzadeh and Howley 2018). These surveys showed that the initial offset metric was difficult to use and open to interpretation and lacked consideration of connectivity and habitat function (Lockhart 2015). All but one of the pilot leaders felt that a voluntary system undermined biodiversity offsetting, especially for lower value habitats (Baker *et al.* 2016). Timing of offset duration was also a contentious issue. Sullivan and Hannis (2015) noted that many landowners viewed in-perpetuity offsetting as fundamentally wrong: ‘we suspect that many landowners will see little that is attractive about dedicating land in perpetuity. They might as well sell it.’ In terms of management, it was felt that over time offsetting could potentially lead to a reduction in time for planning, but this would be marginal compared to the current system (Baker *et al.* 2016). A key concern moving forward was the lack of established legal compliance within the planning sector to protect biodiversity (Taherzadeh and Howley 2018). Similar findings were found when interviewing stakeholders on how offsetting could be implemented in Sweden (Blicharska *et al.* 2022). Stakeholders emphasised the need for transparency when implementing standards, routines and control systems to ensure consistency in how similar cases were handled across the country.

There is a clear divergence in two areas of offsetting when comparing wetland mitigation banking in the USA with the development of biodiversity offsets in the UK: mandatory trading and in-perpetuity offsetting. Both of these relate to the incentives facing developers in needing to purchase offset credits and bankers/offset suppliers wishing to enter the market. We now discuss these in detail within sections 3.2 and 3.3. A consistent theme across the case study analysis is the role of monitoring and enforcement, which we discuss in detail within section 3.4

## In perpetuity offsetting

Conservationists argue that offset sites should be valid in perpetuity to ensure no net loss, or at the very least offset gains should last as long as development impacts (Pilgrim and Bennun 2014; Regnery *et al.* 2013). Evidence from ecological-economic modelling of biodiversity offsets in Alberta (Weber *et al.* 2015) demonstrates that in highly fragmented agricultural landscapes (such as the UK) permanent offsets should be desirable due to the lack of substitutability between sites and that temporary offset sites are not feasible in these landscapes for securing no net loss of biodiversity.

However, what has emerged from the literature review is that a significant barrier to the implementation of offsets in European countries is the length of time that an offset site should be protected. Land managers and regulatory bodies have agreed on significantly shorter timescales (potentially 25 years). In a review of offsetting in the UK, many stakeholders argued that “in perpetuity” is un-attractive for many landowners and making this a requirement could potentially limit the number of offset providers entering the market with many unwilling to permanently give up agricultural and development rights on their land (Sullivan and Hannis 2015). Blicharska et al (2022) used interviews and a workshop to investigate how offsetting could be implemented in Sweden. Their findings draw parrels with current difficulties in UK offsetting around the length of time an offset site should be protected. Landowners are more agreeable to temporary offsets but not those that are protected in perpetuity. As a result, there is a lack of land availability for undertaking offsetting.

In contrast, land use restrictions through conservation easements, coupled with long term management plans and appropriate financial instruments are accepted in both wetland mitigation banking and species conservation in the USA (White *et al* 2021). It was not clear from the literature the main reasons behind the differences in acceptability between the two policies.

## A mandatory offset market?

Evidence to date suggests a mandatory system would be more successful with all developers needing to purchase credits, thus incentivizing landowners to set aside more land to offsetting and create a larger market. Voluntary markets often lead to difficulties in obtaining information about potential buyers and sellers (Wissel and Watzold 2010). In the UK, Defra has been explicit in their preference for voluntary based offsetting, with developers able to choose the compensation option they prefer, a preference backed by the housing developers themselves (Lockhart 2015; Sullivan and Hannis 2015). In contrast, stakeholders from environmental and green business groups emphasised the need for the UK market to be mandatory. Their argument stemmed from concerns that developers will opt against the extra cost of offsets, resulting in a low demand for credits. Resultant low demand would also inhibit supply, impair market efficiencies and result in a lack of a level regulatory playing field (Lockhart 2015; Sullivan and Hannis 2015). Indeed, the UK biodiversity offset pilots failed to deliver a single offset scheme and subsequent analysis suggests that this was due to the scheme being voluntary (Baker et al 2016), alongside the bargaining power of the developers (Carver and Sullivan 2017).

From 2020 to 2021 zu Ermgassen *et al* (2021) developed a database on biodiversity net gain assessments that accompanied planning applications in England. Their work shows the continued preference for developers to deliver their own compensation. There is little incentive for developers to choose to purchase offsets from elsewhere. A key concern is that local authorities do not have the capacity to ensure the delivery of developer-led compensation through regular monitoring and enforcement activities. A similar story is found within the French application of the mitigation hierarchy (Gelot and Bigard 2021). Analysing 2588 development projects, the authors examined i) the characteristics of prescribed mitigation measures in terms of their size, number, target area and layout; ii) their actual implementation in the field, through a spatial analysis. Results showed that many developers ignore opportunities to avoid impacts and skip to reduction and offsetting. As with the UK case, there is a lack of funding for regulatory agencies to adequately monitor and enforce compensatory mitigation and the authors question whether the current policy is achieving no net loss.

In contrast, offsetting using wetland mitigation banking is compulsory in the USA. A significant change in the wetland mitigation banking market was the introduction of the 2008 Mitigation Rule. The rule established a clear preference for off-site compensatory mitigation over on-site mitigation and compensatory mitigation in advance of permitted impacts by commercial mitigation banks (Stephenson and Tutko 2018). The aim of this rule was to reduce mitigation outcome uncertainty by encouraging the development of banks over alternative mitigation options, and thus reduce the financial risk on bank providers (BenDor and Riggsbee 2011). This resulted in a significant increase in the purchase of credits and demonstrates how a regulatory change placing emphasis on mitigation banking above other compensatory measures can have a significant positive benefit on market activity (Figure 2).

Figure 2: Declining use of on and offsite mitigation permittee responsible mitigation (3155 in 2008 to 1437 in 2014) and increasing use of mitigation bank credits (1602 in 2008 to a high of 2113 in 2012 and 2006 in 2014) following implementation of the 2008 Mitigation Rule. Data adapted from Pages 77 – 81 of the [reference needed].

There is however a recognition that developers may not always be able secure credits due to a lack of investment incentive for mitigation bankers. However, rather than undertaking their own compensatory on-site migration in-lieu fee programs serve as a secondary compensatory mitigation option. ILF programs accept fees for permitted impacts and then construct compensatory mitigation projects after sufficient fee revenue has been collected. This measure has been shown to be appropriate in areas where developments are too small or infrequent to stimulate adequate private investment in mitigation banks and to date evidence from the USA suggests that this programme does not compete with commercial banking (Stephenson and Tutko 2018).

Add a concluding paragraph to sum up why voluntary offsetting is a significant barrier!

## Contract design: monitoring and enforcement

A key practical step in the implementation of offsetting is the contract design for the developer and the offset provider. The contract includes the explicit definition of the credits and number of credits required and the timescale for which this should be secured. The role of the regulatory agency is to ensure both parties comply with this contract through follow up monitoring and enforcement.

From a theoretical perspective, it could be argued that neither buyers nor sellers in the offset market are quality conscious, but both are price-conscious. As a result, both parties have an incentive for quick and cheap mitigation and as a result trade regulators need to monitor schemes to ensure compliance (Hallwood 2007; Wainger *et al.* 2010). And in practice, early experience of the USA wetland mitigation banking market showed that schemes suffered from low rates of compliance, despite agreed conditions before implementation (see Brown and Lant 1999; Brown and Veneman 2001; Reiss *et al.* 2009).

Evidence from the literature suggests that developers have significantly greater bargaining power in the design of contracts than the potential offset providers. At the contractual design stage, there is evidence of developers placing downward pressure on compensatory requirements. Vaissière *et al.* (2017) visited 20 mitigation banks and interviewed 54 stakeholders within the Florida wetland mitigation banking market to understand bargaining between developers and offset providers. Developers were actively looking to fulfil their mitigation requirements at the lowest cost and minimise the number of credits they need to purchase. Similar evidence was found when reviewing the UK biodiversity offset pilots (2012 - 2014). Carver and Sullivan (2017) found evidence that firms successfully adjusted the biodiversity metric downward, resulting in not only cheaper compensation for the developers but a potential loss in biodiversity. In one case, the development firm negotiated a reduction in biodiversity offset compensation from £300,000 to £90,000, with the baseline ecological recalculated from 48.68 to 25.52 units. The developers argued the initial cost would restrict the financial viability of the development. Housebuilding is a priority in England, and there are clear tensions between the need to free up land for construction and what developers are prepared to spend on mitigation (Lockhart 2015).

Interviews with wetland mitigation bankers in Florida highlighted the financial risk in developing a bank with significant upfront costs required and a return on investment that may take several years (Vaissière *et al.* 2017). As a result, offset providers may push the negotiations toward an early release of credits in order to obtain revenue more quickly and also aim to have the number credits they can sell uplifted (resulting in ecological uplift). Evidence from the wetland mitigation banking market has shown that when the payment for early release is too high, this incentivizes bankers to reduce their optimal level of effort on the project, potentially leading to sub-optimal restoration (BenDor *et al.* 2014). Thus advanced payment is a fine balancing act: too high and there is an incentive for bankers to underproduce on the restoration effort; too low and not enough bankers will enter the market.

Further empirical analysis has also shown that when the scale of trading and credit release schedules are considered simultaneously, market entry is primarily related to regional geography and regional economic growth (construction rates), i.e., demand for offsets and not whether banks are permitted to release credits early (BenDor *et al.* 2011).

Despite continued calls within the literature that state that monitoring and enforcement is essential to ensure that offset banks deliver the outcomes intended (e.g., Doka *et al* 2022; Vaissière *et al* 2017; Hallwood 2007) there is a significant lack of evidence of this monitoring being undertaken (Bull *et al.* 2018; zu Ermgassen *et al.* 2021). Assessment of the availability and transparency (clear, up to date, and easily accessible information) of data on offset projects implemented under a no net loss objective for France, Germany, the Netherlands and Sweden showed that comprehensive information on offset projects is not yet systematically collated, digitised and disseminated on a national scale; and cannot be accessed remotely (Bull *et al.* 2018).

Within the UK there is little information available on existing compensatory measures (zu Ermgassen *et al.* 2021). Local authorities do not have the capacity to ensure the delivery of developer-owned compensation through regular monitoring. This makes it all the more likely that the same local authorities will struggle to undertake the monitoring and enforcement needed to ensure delivery in a market for biodiversity offsets (Sullivan and Hannis 2015). A similar story is found within the French application of the mitigation hierarchy (Gelot and Bigard 2021). There is a lack of funding for regulatory agencies to adequately monitor and enforce compensatory mitigation and the authors question whether the current policy is achieving no net loss. Indeed, it is possible that the high costs of enforcement associated with offset markets may be so great as to negate any social surplus created through trading in mitigation credits (Hallwood 2007).

The UK, and other countries currently developing offset projects could take some lessons from the USA wetland mitigation banking market. Credit sales within the USA wetland mitigation bank market can be tracked via the Regulatory In-Lieu Fee and Bank Information Tracking System (RIBITS) and it is this information that has formed the case study analyses previously discussed in this paper. Ecological and financial risks within these markets are also mitigated through contractual obligations (Levrel *et al.* 2017). These obligations include:

* Long-term management of the site requires minimal maintenance with very few intensive cost engineering measures.
* The creation of a long-term stewardship fund that will be transferred to a local organisation (NGO) or a public agency to manage the site after the mitigation bank reaches its ecological goals.
* Designation of conservation easements to protect a bank site from any development projects.
* Performance bonds applied during the construction of the restoration site. This is a financial guarantee against any failure to observe the terms of the contract. If a mitigation banker goes bankrupt, a third party associated with the performance bond is obliged to release money from the bond to any other body responsible for overseeing the mitigation bank project. In this way, regulators can carry out the necessary actions to complete the implementation of the mitigation bank, or at least to maintain the ecological lift that has been sold through credits to compensate for impacts.

## Offset metrics and multipliers

The choice of the metric and subsequently the multipliers applied to trades are critical in the design and successful implementation of markets for biodiversity offsets. The metric is the unit of currency for what will be traded and the choice varies from relatively simple (e.g. habitat or a single species) to complex (e.g. ecological functioning or ecosystem services). The multiplier or trading ratio is applied to account for ecological differences between the offset bank and the development site. How many credits the developer needs to purchase is dependent on the ratio of ecological benefits between the offset site and the developed site (McKenney and Kiesecker 2010). Additionally, trading ratios are used to account for uncertainties in the definition and measurement of biodiversity and the discount rate for future gains (Moilanen *et al*. 2009; Bull *et al*. 2013). There are a large number of papers that discuss and review the choice of metrics and multipliers from the ecological perspective that we do not include in this review as the trading and incentivization aspect is not discussed (see Robertson 2004; McCarthy *et al.* 2004; Quetier and Lavorel 2011; Habib *et al.* 2013; Maron *et al.* 2012a; Maron *et al.* 2012b; Maron *et al*. 2016; Gibbons *et al*. 2016).

Within our literature review, we identified 10 papers that explicitly discussed the choice of the offset metric and multipliers and their effect on offset trading. The majority of these were from the environmental economics literature and applied ecological-economic models (optimization and agent-based models) to analyse how alternative metrics and trading ratios influenced economic outcomes (the number of participants in the market and subsequently the number of trades) and ecological outcomes (the achievement of no net loss or net gain) (Heal 2003; Doherty *et* al 2010; Doyle and Yates 2010; Kangas and Ollikainen 2019; Simpson *et al.* 2021a; Simpson *et al.* 2021b; Simpson *et al.* 2022). Optimisation approaches use mathematical programming methods to simulate what decisions land managers should make based on a vector of prices, costs and subsidy rates, and then relate these land management decisions to modelled ecological outcomes. Or, the model can be run backwards to trace out the effects on farm incomes of increasing ecological targets. Agent-based models use simple rules for the behaviour of multiple land managers (“agents”) in a landscape, and then relate these to ecological outcomes using regression-based ecological models.

Fernandez and Karp (1998) provided one of the first economic analyses of wetland banking in the USA. The ecological component of the model describes the uncertain evolution of wetlands using the current state of the habitat, investment in conservation and restoration. The economic component is an optimization problem with the developer choosing an optimal level of investment in restoration and also deciding when to cash in the credits. Results showed that the investor has little incentive to restore the wetlands at a level close to the social optimum. The analysis also revealed that the highest level of restoration would take place when there is a reduction in restoration costs, an increase in biological uncertainty and a rise in the credit price.

A key finding in several papers has been that using a single metric to capture multiple aspects of biodiversity (such as habitat, species or ecosystem functioning) will ultimately fail to protect the aspects of biodiversity not explicitly defined in the metric. Ecological-economic of stream ecosystem service markets in the USA showed that using a single metric to capture ecosystem functioning resulted in the loss of crucial ecosystem functions (Doyle and Yates 2010). Drechsler (2021) used a simulation model to analyse the strengths and weaknesses of a credit that bundles two ecosystem services. The results showed that it is likely that there would be a loss in one of the ecosystem services, despite an overall no net loss being achieved. This result was driven by heterogeneity in the value of land parcels: where an ecosystem service is positively correlated with high-value land parcels (for development) this service will likely decline, as the least profitable land parcels tend to be conserved. This result was also found by Simpson *et al.* (2022) within their simulated market of offsetting in England. It was found that neither the habitat nor species metric adequately captured the indirect benefits of offsetting on related habitats or species. What these papers showed was that within a landscape, the underlying species and habitat distributions when layered with cost data on the supply and demand of credits result in very different landscape outcomes depending on the metric chosen (Kangas and Ollikainen 2019; Simpson *et al.* 2022). These results can be compared with the case study analysis of species conservation banks in Florida by Grimm (2021). The focus of species banking is to conserve protected species under the US Endangered Species Act. However, the metric used in the programme is acreage (not species) and the case study analysis shows that banks ‘overcompensate’ by providing additional acreage but that there is no link to species conservation. This adds further evidence to the already well-developed concerns in the ecological literature that a broad no net loss or no net gain of biodiversity metric does not capture indirect benefits of offsetting on other non-target species (Maron *et al*. 2012; Bull *et al.* 2014; zu Ermgassen 2019; Cristescu *et al.* 2013; Marshall *et al.* 2020a). Ultimately, ecological and economic literature shows that choosing to focus on a single indicator species will not deliver multiple target outcomes for biodiversity (Armsworth *et al.* 2012).

Modelling has also been applied to understand the impacts of the multipliers on ecological and economic outcomes (Bonds and Pompe 2003; BenDor 2009; Laitila *et al.* 2014). System dynamics models of wetland loss and restoration in the USA have shown that even low mitigation failure rates and short delays in compensation can have high costs to society in terms of lost wetland functionality (BenDor 2009). As a result, a multiplier should be employed to reduce uncertainty and losses. There is a balance between too low a multiplier that has no benefits ecologically and too high a multiplier which results in costly credits. Further research has shown that as the duration of the offset shortens, the minimum multiplier should increase (Laitila *et al.* 2014).

From a policy perspective, analysis of stakeholder attitudes towards biodiversity offsetting in England has shown recurring concerns over the lack of evidence that determines the multiplier applied in the trades n (Taherzadeh and Howley 2018). Within policy frameworks, there is often a blind spot on the social effects of offset policies, for example, public preferences for what type of habitats and species should be offset and at what scales that it should take place. Within this review, we identified one choice experiment undertaken in 2012 that assessed public preferences for compensatory mitigation of a proposed new mining operation in New South Wales (Australia). The results showed that the public preference was for a 4:1 trading ratio to reflect how many hectares of offset action would be required to be equivalent to the community value lost from a hectare of cleared endangered ecological communities. We did not find further examples of this within the literature.

Overall, whilst there are significant concerns arising from the ecological literature on the choice of offsets and multipliers and evidence from the environmental-economics literature that no single metric can deliver economically and ecologically beneficial outcomes, this does not seem to be the main driver of the lack of uptake in offsetting by developers or lack of offset creation. Indeed, practical considerations regarding timescales, geographic trading scale, regulatory capacity and bargaining powers seem to be the main obstacles.

## Geographic Scale

The ecological and economic outcomes of an offset market are also influenced by the scale, by which we mean the area over which developers and offset providers can trade. Large trading regions increase potential demand for credits whilst small service areas reduce demand but large trading regions allow conservation to be distant from impacts (BenDor *et al* 2011). Similar results were found in ecological-economic simulations of offset market in England that showed that larger trading areas benefit developers by placing downward pressure on credit price, thus benefitting developers. In contrast, smaller trading areas were more beneficial ecologically by reducing the distance between impact and offset site, but also increasing the credit by price and therefore reducing credit demand, and therefore less negative impacts on biodiversity (Simpson *et al*. 2021). Conservationists have long argued that large trading regions run the risk of offsets being located far from the original site leading to changes in the functional quality and scale of habitats. Reviews of the USA wetland mitigation banking markets have shown that in some states there are high levels of cross watershed mitigation, resulting in changes to function quality and scale of wetlands (BenDor and Brozović 2007; Kettlewell *et al*. 2008). However, isolated mitigation projects are more likely to fail when compared to a project that is incorporated into a larger, ecosystem-based conservation bank or regional conservation plan (Crooks and Ledoux 2000). Furthermore, economic analysis of offset markets has shown that in developing mitigation banks, economies of scale may be realized from large, off-site mitigation banks with one large-scale wetland restoration site may cost less and provide more environmental benefit than several small restoration sites (Bonds and Pompe 2003). Ultimately, the fundamental constraints created by the regional landscapes and economic dynamics drive market demand (BenDor *et al* 2011).

## The role of transaction costs

Transaction costs are expenses incurred when buying or selling a good or service and are a key feature of markets for biodiversity offsets. Developers incur transaction costs through the time and resources expended first to understand what an offset was and then in determining what to propose as an offset for their impact. The effect on transaction costs depends on the guidance available to developers to inform how many offset credits they need to purchase. Identifying an offset supplier also incurs transaction costs, as well as the costs incurred during negotiation to finalise the offset arrangement. For suppliers, the main transaction costs arise from the determining what they can contribute as an offset. Scheme administrators also incur transaction costs in terms of the time and resources needed to determine whether an offset is suitable; the processing, negotiating and re-negotiating of offset proposals and also the follow up monitoring and enforcement. For offsets to be considered cost-effective, these transaction costs need to be less than the benefits from trade (Coggan *et al* 2013). Despite the recognition of transaction costs as a key feature of offset markets (and markets more broadly) there is a lack of empirical literature on the subject. Several papers make reference as to how transaction costs can be reduced e.g. generating ways for offset buyers and sellers to quickly access accurate ecological information, supporting the emergence and use of information specialists such as brokers, and the development of a database of suppliers that is publicly available to all developers (Bonds and Pompe 2003; Coggan *et al* 2013; Vaissière *et* al 2017; Kangas and Ollikainen 2019). However, there appear to be no papers that formally analyse transaction cost data within biodiversity offset markets.

## A lack of cost data

In defining the market for offsetting the key questions are who are the participants? The credit market depends on the rate of land-use change caused by development and the difference in opportunity costs of conservation for land patches (van Teeffelen *et al.* 2014). The potential players are local landowners who will provide land for offset schemes; developers looking to purchase land for development and also offset their impacts on biodiversity and also environmental NGOs and charities who may also wish to buy offsets to limit land for development and secure conservation gains. In many situations, compensatory measures are likely to be undertaken on agricultural land, with a loss in agricultural income.

In undertaking this literature review, there is a lack of research focussed on incentives for landowners to participate in such schemes. One notable research paper focussed on farmers' preferences for implementing compensation measures and how acceptance could be improved in Germany (Sponagel *et al* 2021). Using a discrete choice experiment, 209 farmers were surveyed to establish their preferences on the protection of land in perpetuity, the maintenance period for compensatory measures and the type of compensation. The results showed that the type of compensation measure and the monetary value received were both critical in farmer acceptance: the complete conversion of arable land to grassland was the least accepted measure, with farmers preferring a compensatory action that would still allow them to undertake cropping on some areas of the farm. Farmers were also not in favour of protecting their land in perpetuity (a view we know is shared by farmers in other European countries) and instead opted for a maintenance period of 25 years.

A barrier to the development of BDO in England has been a shortage of offset sites from which to purchase biodiversity offset credits, as confirmed in DEFRA's pilot evaluation report (Baker *et al.* 2014). Analysis of written submissions to the UK Parliament׳s Environmental Audit Committee׳s 2013 Inquiry into Biodiversity Offsetting in England highlights those bodies representing agricultural landowners require offset credits to be priced higher than simply income forgone, with offset providers should profit financially from their participation in the market (Sullivan and Hannis 2015). However, there is a distinct lack of available data on restoration costs, transaction costs and land prices for offset implementation in Europe (Simpson *et al* 2021, 2022; Kangas and Ollikainen 2019). This is challenging from a research perspective when considering how best to design an offset market for both ecological and economic effectiveness, but also in incentivising landowners to participate in a market. Simpson *et al.* (2021, 2022) attempt to value habitat and species-based offsets in the UK using agricultural gross margin data acting as a proxy for a farmer’s minimum willingness to accept a change in land use from agriculture to offsetting and compare this with developers’ maximum willingness to pay for a credit as a function of land value for housing and the number of credits required. However, critically this does not include restoration cost data and transaction costs data. Kangas and Ollikainen (2019) needed to rely on expert assessments and assumptions to develop their economic model of offsetting in Finland.

In Australia, economic valuation of ecosystem service markets has been undertaken at the landscape scale to consider how sustainable farming can deliver carbon sequestration, agricultural production, water, biodiversity and timber production (Baral *et al.* 2014). The analysis showed that under current prices, agricultural production remained the most economically beneficial options for farmers and that the income from carbon sequestration and biodiversity conservation would need to be considerably higher than current levels in order to justify focusing management of this landscape on ecological outcomes.

Even within the well-established USA wetland mitigation banking market it is difficult to source reliable cost data. Mitigation bankers are often reluctant to reveal bank cost information (BenDor *et al* 2011). The most recent analysis by Poudel and Pokharel (2021) costed 26 habitat conservation banks in California, compiling their cost and revenue data. The average annual cost of operating a bank was $42 /acre (or £33 GBP per acre) and the average credit price was $6014 /acre (or £6014 GBP per acre). The authors found that financial performance depended on the location of the project and the size of the landholding, with larger offset banks generated generating returns for the investment spent. Furthermore, they found that banks that received large upfront payments (before the restoration was established) were more likely to not meet the conservation goals and had poor follow up management of the site.

For biodiversity offsetting, reliable price signals reduce the uncertainty on investor returns from development and in turn reduce development costs (Salzman and Ruhl 2004; Wissel and Watzold 2010).

# What can we learn from other incentive-based conservation schemes?

BRING IN THE PAPERS IDENTIFIED BY JOE HERE

1. **Other future research directions (to be written)**

*Ecosystem service markets:* Should we be including the full suite of ecosystem service values? “Ecosystem service valuation must be incorporated into wetlands mitigation banking decision making at two critical junctures: the wetlands assessment stage and the wetlands trading stage. To fully capture ecosystem service values, the assessment method must consistently define the services and incorporate measurements of their value both for the wetlands to be lost and for those wetlands used for mitigation (Ruhl and Gregg 2001)

*Financial feasibility:* In particular the following issues need to be considered: What are the values of the marine priority habitats affected by residual impacts? How much will a credit cost? What is the cost of creating the different offsetting measures? How will credits take account of differences in the habitat type impacted and the proposed offset? What effect will additional costs associated with purchasing credits have on the economic viability of the development types of interest to The Crown Estate? What is the value of the role of offsetting in ‘unblocking’ the consents regime leading to a speedier licensing outcome? What is the interest level in financial institutions in developing a credit trading system? (Crown Estate Marine Research Report)

*Landscape scale modelling for biodiversity offset markets:* (Williams et al 2012) Therefore, we need ways to comprehensively value alternative landscape-scale outcomes, which can be linked (where necessary) with a multi-round tender to deliver cost-effective payments for conservation services in the right place at the right time. To be effective, a landscape ecological metric requires a more integrated model of biodiversity persistence or habitat integrity to account for values associated with the type, extent and condition of habitats (representing biodiversity at the site-level, between sites and across whole regions); how these attributes change over time and with management; and the minimum prerequisites for a broad range of species to persist in situ (McCarthy et al., 2004).

1. <https://ribits.ops.usace.army.mil/ords/f?p=107:158:15110915550763::NO> [↑](#footnote-ref-1)