Q Open, 2021, **1**, 1–19
DOI: 10.1093/qopen/qoab004
Advance access publication date: 26 February 2021
Article

Incentivising biodiversity net gain with an offset market

Katherine Simpson[1], Nick Hanley [1,*], Paul Armsworth[2], Frans de Vries[3] and Martin Dallimer[4]

[1]University of Glasgow, UK
[2]University of Tennessee Knoxville, USA
[3]University of Stirling, UK
[4]University of Leeds, UK

*Corresponding author: Institute of Biodiversity, Animal Health and Comparative Medicine, University of Glasgow, G12 8QQ Glasgow, UK. E-mail: Nicholas.Hanley@glasgow.ac.uk

Received: December 14, 2020. **Accepted:** February 12, 2021

Abstract

Most programmes that incentivise the supply of public goods such as biodiversity conservation on private land in Europe are financed through the public purse. However, new ideas for how to fund biodiversity conservation are urgently needed, given recent reviews of the poor state of global biodiversity. In this paper, we investigate the use of private funding for biodiversity conservation through an offset market. The environmental objective is to increase some measure of biodiversity in a region ('net gain') despite the loss of land for new housing. Farmers create biodiversity credits by changing their land management and then sell these credits to housing developers who are required to more than offset the impacts of new housing development on a specific indicator of biodiversity. Combining an economic model of market operation with an ecological model linking land management to bird populations, we examine the operation, costs, and biodiversity impacts of such a (hypothetical) market as the target level of net gain is increased. A general result is established for the impacts on price and quantity in the offset market as the net gain target is made more ambitious. For a case-study site in Scotland, we find that as the net gain target is increased, the number of offsets traded in equilibrium falls, as does the market-clearing offset price. Changes in the spatial pattern of gains and losses in our biodiversity index also occur as the net gain target is raised.

Keywords: offset markets, net gain, biodiversity economics
JEL codes: Q01, Q21, Q24, Q57

1 Introduction

Approximately 75 per cent of global land has been 'significantly altered' by human development, largely driven by land-use change for infrastructure and agricultural production. These processes are predicted to continue, with nearly a million species thought to be at risk of extinction in the coming decades if no remedial action is taken (IPBES 2019). While some call for over half the planet to be given over to biodiversity conservation (e.g. Dinerstein *et al.* 2020), such an approach inevitably incurs significant economic and social costs. Indeed, halting land conversion and agricultural production to the extent required to address biodiversity losses may mean that it is impossible to meet a suite of UN Sustainability Development Goals (United Nations 2018; IPBES 2019; Nature 2020). In place of restricting development completely, new approaches need to be applied that give governments, developers, regulators, and wider society new tools to help reduce the negative impacts of development pressures on the natural world (Simmonds *et al.* 2020).

One such approach is biodiversity offsetting. Biodiversity offsetting aims to provide 'measurable conservation outcomes resulting from actions designed to compensate for significant residual adverse biodiversity impacts arising from project development' (BBOP 2009). To date, around 3,000 offset projects have been recorded worldwide covering at least 153,670 km^2, with the greatest number of compensation projects taking place in Brazil and Mexico (Bull and Strange 2018). A widely accepted view is that 'no net loss' should be the minimum standard for safeguarding biodiversity in the face of development impacts (Maron *et al.* 2020). More recently, countries are beginning to explore policies that focus on net gain—a net positive impact on some indicator of biodiversity (Gibbons and Lindenmayer 2007; Maron *et al.* 2018). Net gain requires actions that ensure recreated or restored habitats exceed those lost in terms of potential biodiversity outcomes (i.e. gains outweighing losses in some agreed metric) (CIEEM, CIRIA, and IEMA 2016). Net gain can be achieved in two ways: (1) by over-compensating directly for the loss in biodiversity affected by development or (2) by ensuring no net loss in the directly impacted biodiversity and then providing additional gains in other biodiversity values, known as 'out of kind' compensation (Bull and Brownlie 2017). Critical to both of these is the idea of additionality: only those actions that would have not otherwise occurred should be counted towards the creation of a biodiversity offset credit (Laitila *et al.* 2014).

A number of questions arise when exploring the implementation of biodiversity net gain. First, what should be the level of net gain delivered? This is an emerging debate within the ecological community at a global scale (see Bull and Brownlie 2017; Weissgerber *et al.* 2019; Maron *et al.* 2020 for a detailed discussion). Maron *et al.* (2020) argue that in countries where ecosystems are most severely depleted, net gain is essential. However, in rare circumstances, a managed net loss should be allowed, where there is the greatest human development need and where natural ecosystems remain extensive (Maron *et al.* 2020). The second question is who will deliver the net gain in biodiversity and at what economic cost. Within the context of housing and infrastructure development, a shift from a no net loss of biodiversity to some level of net gain creates additional demand for schemes that restore and recreate habitats to mitigate development impacts. One option to meet this increased demand is through a market for biodiversity offsets. Markets are created when multiple buyers and sellers of biodiversity offsets interact with each other through a trading process. This creates a setting in which landowners can choose to manage land for conservation and generate offset credits. These credits can then be sold to a developer who is required to mitigate development impacts on some specific biodiversity metric. Such trades can be facilitated by an offset bank, which collects offers from sellers and makes these available to potential buyers. Buyers will not offer more for a credit than the value to them of land for development, and sellers will require no less in payment than the opportunity costs to them of creating offsets (Needham *et al.* 2019).

There is an expectation that a change in the policy agenda from no net loss to a net gain will affect the demand for offsets, and thus the functioning of a market for biodiversity offsets. Impacts likely include those on the prices at which credits are sold, the cost to developers, the gains to landowners who supply credits, and the resulting ecological landscape. This paper contributes to the literature by examining the economic and ecological impacts of increasing net gain requirements on a market for biodiversity offsets. We are not aware of any papers that have done this before. Using an integrated ecological–economic modelling approach, we examine the following important questions:

1. How does the equilibrium market-clearing price and quantity of offsets vary according to an increasing requirement for biodiversity net gain?

2. What is the effect on housing developers of a move from no net loss to net gain?
3. What are the ecological impacts of having no offset policy compared with various net gain scenarios, and how do these impacts vary spatially?

To answer these questions, we first develop a conceptual model of a biodiversity offset market that compares the demand for offsets under (1) a policy target of no net loss with (2) a policy target of biodiversity net gain. We then develop an empirical application of this conceptual model, which includes spatially explicit biodiversity offset supply and demand curves under a range of no net loss and net gain policies. These supply and demand curves capture the spatial variations in the costs of supplying biodiversity offsets, which depend on the relative value of land for agriculture, and the demand for offsets, which depend on the value of new housing developments across the landscape and the net gain requirement.

We explore this in the context of the UK, as it is the first country in Europe to adopt legislation requiring a net gain in biodiversity for new development projects (infrastructure and housing) (HM Government 2018). It is estimated that the restoration of UK priority habitats will cost an annual average of £97 million per year (Rayment 2019). Whilst some of this can be delivered through the new Environmental Landscape Management scheme—and thus be funded directly by the UK taxpayer—longer term land management measures, including habitat restoration and creation, will require a different incentive structure than current agri-environment contracts.

The remainder of the paper is organised as follows. Section 2 develops our conceptual model of biodiversity net gain. Section 3 provides information on the methods used, including the integrated ecological–economic model and the case-study region. Section 4 provides the results. Section 5 presents the discussion and conclusions.

2 Modelling the biodiversity offset market

2.1 General structure

We consider a region where land can be divided into three possible uses, which are mutually exclusive at any point in time: agriculture, development for new housing, and biodiversity conservation. We assume the land is currently owned and managed by farmers, while there are developers who wish to acquire land for housing development. We further assume that both farmers and developers are price-takers in their respective output markets as well as the market for offsets, i.e. no individual in either group has market power in either the offset market or the housing market.

In the first instance, for a land parcel to be developed, the developer must hold the relevant quantity of offsets, q, to satisfy a no net loss of biodiversity policy, that is the developer must buy one offset if it wants to develop one parcel of land. This requirement generates a demand for offset credits from developers, $D(q)$. Ranking parcels of land that could be developed according to their expected housing value from highest to lowest yields a downward-sloping demand curve for offsets. Here, we assume that heterogeneity in reservation prices stems from differences in housing development rents (or house prices) across the landscape. Due to our assumption that the housing market is competitive, the reservation price for an offset that will allow development of a particular land parcel equals the net profit available from developing that land parcel. In equilibrium, each developer will choose to buy a quantity of offsets that equates their individual demand curve with the market price of offsets.

Offsets are supplied by farmers who choose to convert their existing agricultural land to a conservation land management scheme that benefits biodiversity and thus generates a supply of offset credits, $S(q)$. We assume that for a farmer to supply offset credits, they must be compensated at minimum by the offset market for the opportunity costs of the foregone agricultural profits plus any associated restoration costs. Across the landscape, agricultural profits per land parcel may vary due to variations in agricultural productivity. This allows

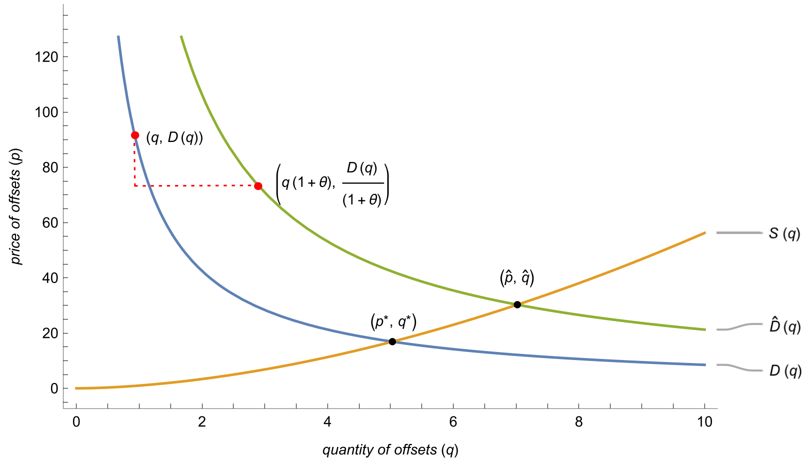

Figure 1. Demand shift from no net loss to net gain policy and equilibrium price–quantities.

us to rank all agricultural land as a continuous, upward-sloping supply (or marginal cost) curve describing how many new offsets will be created for a given price, p. Each farmer maximises profits by choosing to supply the quantity of offsets that equates the marginal cost of creating new conservation areas (i.e. lost agricultural profit at the margin) with the offset price, p.

2.2 From no net loss to net gain

To keep the model transparent, assume that when the offset market operates under a no net loss biodiversity objective, the developer must buy one offset if it wants to develop one parcel of land, implying that all land is of equal conservation value (this assumption is relaxed in our empirical model). Given the aggregate demand and supply function introduced previously, the offset market will be in equilibrium when $D(q) = S(q)$, generating an equilibrium quantity of offsets, q^*, and equilibrium market price, p^* (see Fig. 1).

A no net loss policy has the objective of preventing any decrease in some specified biodiversity metric as a result of development. Taking the no net loss policy as our baseline, we want to assess the configuration of the offset market if, instead, a net gain policy objective was to be imposed, where 'net gain' means the policy target changes to one of achieving a specified *increase* in the biodiversity metric as a result of development. Changing from a no net loss to a net gain requirement does not affect the agricultural productivity of the land parcels, nor their potential ecological value, and hence does not directly affect the costs to the farmer supplying offset credits. For this reason, changing the net gain target implies no shifting of the offset supply curve. In contrast, as we show below, changing the net gain target has an effect on the demand curve for offsets.

Under a net gain policy objective, a developer needs to purchase a quantity of offsets equal to $(1+\theta)$, where θ is the percentage net gain that is required. To fix ideas, let us consider a given position on the demand curve $(q,\ D(q))$ under the no net loss policy. Under a net gain policy, developments up to and including that point now need to be offset by $q(1+\theta)$ and the relevant developer's reservation price for each offset decreases from $D(q)$ to $D(q)/(1+\theta)$. Thus, relative to the baseline, the changes in the quantity demanded and corresponding reservation price for each offset can be represented by $(q(1+\theta),\ \frac{D(q)}{(1+\theta)})$. Reorganising to express this new demand curve just in terms of a generic quantity of offsets

q, we find that moving to a net gain policy shifts the original demand curve derived under the baseline policy to the new demand curve described by (see Fig. 1)

$$\hat{D}(q) = \frac{D\left(\frac{q}{1+\theta}\right)}{1+\theta}. \tag{1}$$

Since the original supply function is unchanged, the new market equilibrium under a net gain policy entails the equilibrium quantity and price combination $(\hat{q},\ \hat{p})$ that solves (see Fig. 1)

$$\hat{D}(\hat{q}) = S(\hat{q}). \tag{2}$$

As the net gain parameter θ is varied, the equilibrium price–quantity combination will vary, so $(\hat{q},\ \hat{p}) = (\hat{q}(\theta),\ \hat{p}(\theta))$.

This brings us to a point that allows us to determine how the market equilibrium price and quantity of offsets is affected by an increasing biodiversity net gain requirement. Applying implicit differentiation of (2) after substituting in the expression for the new demand curve from (1), we can recover the effect of increasing the net gain requirement θ on the equilibrium quantity of offsets (see Appendix in the Supplementary Material for derivation):

$$\frac{\partial \hat{q}}{\partial \theta} = \frac{\frac{\hat{q}}{1+\theta} D'\left(\frac{\hat{q}}{1+\theta}\right) + D\left(\frac{\hat{q}}{1+\theta}\right)}{D'\left(\frac{\hat{q}}{1+\theta}\right) - S'(\hat{q})(1+\theta)^2} \gtrless 0. \tag{3}$$

The general expression in (3) indicates that the effect of an increase in the net gain requirement is ambiguous and depends on the sign of the numerator (noting that D' is negative and the denominator is negative). However, by focusing on a small change from the market equilibrium in the no net loss baseline (i.e. a small movement towards a net gain policy), we can evaluate the effect of a marginal increase of θ more accurately. That is, evaluating (3) at $(q(\theta),\ \theta) = (q^*, 0)$ yields (see Appendix in the Supplementary Material for derivation and proof):

$$\left.\frac{\partial q}{\partial \theta}\right|(q^*, 0) \gtrless 0 \Leftrightarrow \epsilon \gtrless 1, \tag{4}$$

where $\epsilon = \frac{\partial q/q}{\partial p/p}$ is the price elasticity of demand for offsets. Thus, (4) indicates that the change in demand for offsets due to a marginal increase in the net gain requirement is solely determined by the price elasticity of demand at the no net loss equilibrium. The following proposition summarises our key finding:

Proposition 1. *In competitive markets for biodiversity offsets and development, a biodiversity net gain policy initially increases (decreases) the equilibrium quantity and equilibrium offset price if offset demand at the no net loss equilibrium is inelastic (elastic).*

As reflected in (3), our main theoretical result in Proposition 1 suggests that the effect of an increasing net gain requirement θ on the equilibrium quantity (and price) of offsets is ambiguous, but it allows us to predict the direction of change more precisely depending on the elasticity of demand at the original no net loss baseline. To understand why the effect is ambiguous, it helps to consider how the decision of the marginal developer will be affected. Should the marginal developer proceed with development after a shift to a net gain policy, then they and all other developers to the left of the equilibrium in Fig. 1 will need to purchase additional offsets. On its own, this effect would increase the equilibrium quantity of offsets. However, the developer's maximum willingness to pay for each individual offset also decreases as a result. On its own, this second effect would likely induce the marginal developer to no longer proceed with development. The ambiguity in (4) reflects the interplay between these two effects. Another way to put this finding in perspective is to compare this result with the extreme cases of perfectly inelastic($\epsilon = 0$) and perfectly elastic ($\epsilon = \infty$)

offset demands. When $\epsilon = \infty$ (the demand curve is horizontal), developers face the same rent (profit) value from development everywhere, meaning the demand curve is a horizontal line. When imposing a net gain requirement in this case, each developer's reservation price for an offset decreases uniformly, leading to a lower price and quantity of offsets demanded in equilibrium. In contrast, $\epsilon = 0$ (the demand curve is vertical) implies an infinitely valuable development requiring a given number of offsets under a no net loss policy. Increasing the policy requirement to one of net gain per unit of development would just shift that demand curve to the right, leading to an increase of the equilibrium price and quantity.

In the following section, we develop an empirical version of the above model and use this to test the theoretical prediction summarised in Proposition 1.

3 The empirical approach

For our empirical example, we employ an integrated ecological–economic modelling approach first developed in Needham *et al.* (2020).The model seeks to maximise each landowner's joint profit derived from agriculture and new housing development within a region, subject to a regulatory limit of no net loss or net gain of a single ecological policy target. As a baseline, we take the current land-use structure in the case-study area that is of a fixed size. This area is divided into a number of $i = 1, \ldots, n$ land parcels equalling a size of 100 ha and aligned to the Ordnance Survey British National Grid. Each land parcel is assumed to be managed by a single landowner. Each land parcel can comprise any combination of thirty distinguished land-use types, including agricultural and crop classifications based on the Land Cover and Land Cover Plus Crops Map (Rowland *et al.* 2015).

Our model focuses on a single hypothetical biodiversity policy target: populations of the Northern lapwing (*Vanellus vanellus)*. Lapwing appears on the Red List (species in most urgent need of conservation action) of threatened bird species in the UK (Eaton *et al.* 2015). Lapwing populations have declined by 54 per cent in the past 50 years, partly due to changing farmland management. We use statistical regression on observed lapwing numbers across UK farmland to describe the relationship between the current land use and the current abundance of birds within a land parcel (Needham *et al.* 2020). Our estimate of current lapwing abundance represents the baseline no net loss conservation objective. For the net gain scenarios, the net gain objective is calculated on a parcel by parcel basis, with the baseline abundance simply multiplied by the net gain policy objective. For example, an offset for a parcel with a current abundance of two lapwings would require habitat improvements elsewhere that would support three additional lapwings to be purchased under the 50 per cent net gain scenario.

Lapwing numbers can be increased on land parcels by farmers replacing the current land use on a parcel with an ecologically preferred land management practice, which enables offsets to be generated. We specify this conservation land management practice as improved grassland without livestock grazing. Grassland is more beneficial to lapwing population abundance than the alternative agricultural practices of crop and/or livestock production (Needham *et al.* 2020). To ensure additionality, this change in land management practice in the model can only take place on agricultural land patches currently farmed for crops or livestock. Our model assumes that contracting terms involved in generating offsets are such that the conversion of agricultural land to offset provision will be permanent, that is a farmer cannot switch between agricultural production and offset provision on a given land parcel as time goes forwards.

In contrast to the theoretical model of Section 2, land parcels are heterogeneous in ecological terms: ecological quality or potential varies between parcels, although not within each parcel. This implies that some land parcels can generate more lapwings, equating to a greater number of offsets generated, from a switch from current cropping to the same conservation land management practice. Hence, higher ecological potential land parcels generate more valuable offsets when the land management conversion is undertaken than land parcels with lower ecological potential (lower ability to 'produce' additional lapwings).

We assume that on all agricultural land parcels, the current crop and livestock distribution is at present the most profitable to a farmer, and that switching to an alternative land management practice will result in a loss of profit, incurring an opportunity cost. For each land parcel, we generate the farmer's minimum willingness to accept (WTA) a change from current agricultural land use to the conservation land use (grassland with no grazing). This gives us the minimum unit price at which a farmer would be willing to supply one offset:

$$WTA_i = \frac{\pi_i^1 - \pi_i^2}{\Delta b_i} \;\; (i = 1, \ldots, n), \tag{5}$$

where π_i^1 is the gross margin of the land parcel under the current agricultural land use, π_i^2 is the gross margin of the land parcel under the proposed land management practice (grassland), and $\Delta b_i = b_i^2 - b_i^1$ denotes the increase in the lapwing (b) abundance gained from the binary decision to shift all agricultural land within a parcel to grassland. Gross margins, the difference between crop revenue and variable costs, are taken from Scottish farm management data (Beattie 2019), and are applied to current cropping/livestock use patterns in each parcel as shown by land cover and agricultural census data, adjusted to reflected variations in productivity across space (see the Supplementary Material for more detail). Knowing the farmers' WTA, land parcels can now be ranked according to the offset value they offer in terms of increased lapwing abundance, recovering the analogous supply curve to that shown in Fig. 1.

Let us next consider what shapes the demand side of the market. A developer's demand for offsets is determined by the expected value of the land for new housing development and the requirement to purchase offsets to satisfy the no net loss or net gain policy criteria. For each land parcel, we generate the developers' maximum willingness to pay for a single offset:

$$WTP_i = \frac{r_i - \pi_i^1}{b_i^1\,(1+\theta)} \;\; (i = 1, \ldots, n), \tag{6}$$

where r_i is land rental value of the parcel for housing developement, $\pi_i^1$is the agricultural gross margin of the land parcel under the current land use, b_i^1 is the abundance of lapwings currently supported by the land parcel, and θ is the percentage net gain required under the no net loss or net gain policy.

Knowing the developers' maximum WTP for offsets, each land parcel can now be ranked according to its development value. This reveals which land parcels deliver the most offset profitable housing developments taking into account the offset requirements, and allows us to recover the analogous demand curve for offsets to that shown in Fig. 1. Empirically, we make use of UK land registry data to recover average house prices for each land parcel (since no comprehensive dataset is available on land values for housing). We then assume that 50 per cent of the value of each house sold reflects the value of land, based on information in HM Government (2018). This allows an estimate for each land parcel of the value of land for new housing development, which indicates the value of an offset credit that permits development on each of these parcels (the maximum WTP of developers for offsets).

Having recovered the empirical supply and demand curves for offsets, we are now in a position to determine the price for a single offset at market equilibrium (p^*). At the market equilibrium price, we then determine whether a land parcel remains under current land use, becomes an offset supplier, or is developed for housing. In particular, at each price point, the farmer managing site i will supply offsets if the market offset price $p^* \geq WTA_i$. In the same vein, at each price point, a developer will buy and develop parcel i and purchase offsets to compensate for the loss of lapwing if $p^* \leq WTP_i$.

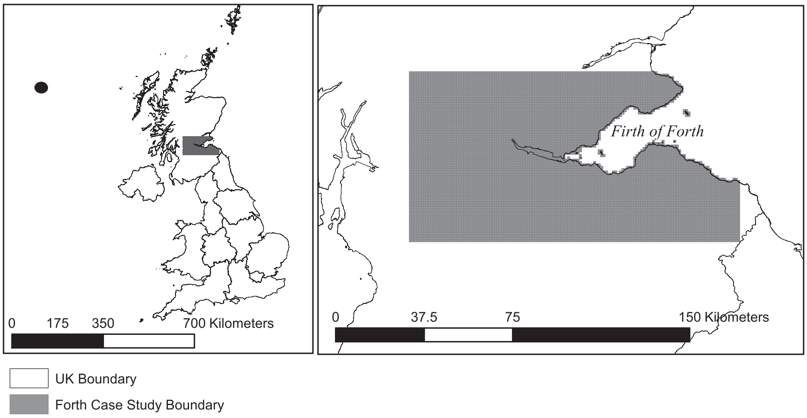

Figure 2. Case-study location.

We examine seven scenarios using our empirical model. Our aim is to compare the market-clearing price for offset credits, the costs to developers of purchasing offsets, and the subsequent economic and ecological impacts market under different levels of net gain and no net loss, compared with the case where no requirement to offset is imposed at all.

- Scenario 1: *Full development on all profitable land parcels with no offsetting.* Under this scenario, all land parcels within the case-study area that have a positive value for housing development are developed. This is the most 'extreme' level of development with no planning regulations imposed, including no regulations on biodiversity loss. This would represent the greatest loss in the ecological policy target (the biggest decline in lapwings).
- Scenario 2: *No Net Loss:* Under this scenario, land parcels can be developed as long as the landowner holds an equal number of offset credits to the number of lapwings being lost. In this case, the net change in lapwings across the region as a result of new housing development should be zero. That is, local losses in lapwings are exactly balanced by gains elsewhere in the region.
- Scenarios 3–8: *Biodiversity Net Gain of* $\theta = 5$ *per cent, 10 per cent, 20 per cent, 30 per cent, 40 per cent, or 50 per cent.* Under this scenario, housing development can only occur so long as sufficient credits are supplied and purchased to increase the lapwing population across the region by the given level of target net gain.

Comparing Scenario 1 with Scenarios 2–8 allows us to explore the ecological impacts of unregulated development compared with different levels of net gain. We can also explore spatially how ecological impacts change across the landscape under different scenarios. Comparing Scenarios 2–8 allows us to examine the effects of increasing levels of net gain on the market-clearing price for biodiversity offsets, and on the economic cost to developers of increasing net gain requirements.

A full account of all data sources used, and how these were processed for use in the model, can be found in the Supplementary Material.

3.1 Case study

We apply our modelling approach to the Forth Valley, central Scotland, UK (Fig. 2).The Forth Valley contains a mosaic of biodiversity-rich habitats from wetlands, marshlands, and heather uplands, some of which are protected through the EU Habitats and Wildlife Birds Directives (92/43/EEC and 2009/147/EC). Biodiversity-rich areas out with these designations face pressure from the growing population within the central belt region for new housing (the case-study area covers the cities of Edinburgh and Stirling), as well as further expansion of the heavy industry, ports, and petrochemical complex of Grangemouth and Rosyth. As such, this provides an ideal example in which to test our market for biodiversity offsets and net gain. Whilst the current UK net gain policy is being pursued by DEFRA through the central UK Government, the Scottish Government have committed themselves to similar policy frameworks through their Environment and Biodiversity Strategies, overseen by the regulatory and advisory bodies SEPA and Scottish Natural Heritage (Scottish Government 2020; Scottish Natural Heritage 2019).

To predict the changes in lapwing in any parcel when it is switched from the current land use to either development or conservation, we estimate a Species Abundance Model (SAM) for the eastern region of the UK that can then be used to predict the current distribution and abundance of the target species for every grid square in the Forth valley based on existing land-use patterns. SAMs generate predictions of abundance for unsampled locations in a study area using existing data from other sample areas, extrapolating this to new areas based on environmental characteristics (Barker *et al.* 2014). The development of these ecological models allowed us to obtain predictions for the abundance of lapwing based on current land use and any land-use change as a result of biodiversity offsetting. The first stage of the modelling process was to produce a multiple regression model to identify which parameters (environmental characteristics) predict current distributions for lapwing across the eastern region of the UK. This allowed us to identify which land management practices are most beneficial and thus serve as our 'preferred' land management practices within the biodiversity offset market. The SAM was then used to predict how the distribution and abundance of lapwing alter as a result of land-use change at a 1-km grid square level across the Forth valley, allowing us to generate the predicted biodiversity offset credits. For more details on this ecological model, see Needham *et al.* (2020).

4 Results

Recall our first research question: how does the equilibrium market-clearing price of offsets vary according to an increasing requirement for biodiversity net gain? Within Proposition 1, we showed that the effect of increasing the net gain requirements on the equilibrium quantity (and price) of offsets is ambiguous. That is, a biodiversity net gain policy initially increases (decreases) the equilibrium quantity and equilibrium offset price if offset demand at the no net loss equilibrium is inelastic (elastic). For our case study, we find that the price elasticity of demand at the no net loss equilibrium is −7.13, i.e. development demand in this system is highly elastic. As a consequence, we find that the price and quantity of offsets traded are highest at the no net loss policy target (Table 1). Under no net loss, the market-clearing price per lapwing offset is £16,433 and 232 trades take place. Both the number of trades and clearing price consistently decline as the net gain requirement increases: at 5 per cent net gain, 217 trades take place at a market-clearing price of £16,083 per lapwing, declining to 169 trades taking place at 50 per cent net gain, with a market-clearing price of £14,137 per lapwing (Fig. 3). As summarised in Section 2, moving from a no net loss to a net gain requirement increases the demand for credits for each developer, with more offsets required per land parcel for development to be allowed to take place. However, this reduces their willingness to pay for a single offset, resulting in the demand curve for offsets shifting downwards and to the right. This is shown in Fig. 4, where the demand curve is shown for the no net loss and 50 per cent net gain policies.

Our second set of results considers the economic costs to developers of no net loss and net gain policies in comparison to development without biodiversity regulation (Table 1). Recall Scenario 1, where all land parcels that are profitable for housing are developed regardless of their biodiversity value, here 4,066 land parcels are developed. In comparison, under

Table 1. Overview of the change in economic costs as the biodiversity net gain requirement increases from no net loss to 50 per cent net gain.

	Scenario 1: no regulation	Scenario 2: no net loss	Scenario 3: 5% net gain	Scenario 4: 10% net gain	Scenario 5: 20% net gain	Scenario 6: 30% net gain	Scenario 7: 40% net gain	Scenario 8: 50% net gain
Market-clearing price for an offset	n/a	16,432	16,083	15,876	15,247	14,880	14,402	14,137
Land parcels developed for housing	4,066	1,042	1,018	999	966	930	911	883
Cost of offset purchase to house builders (million)	n/a	£3.80	£3.31	£3.06	£2.62	£2.16	£1.86	£1.59
Change in the number of lapwings	−6,991	0	+11	+19	+34	+45	+51	+57
Cost of foregone housing development (million)	0	£35.33	£35.75	£35.98	£36.36	£36.85	£37.17	£37.52

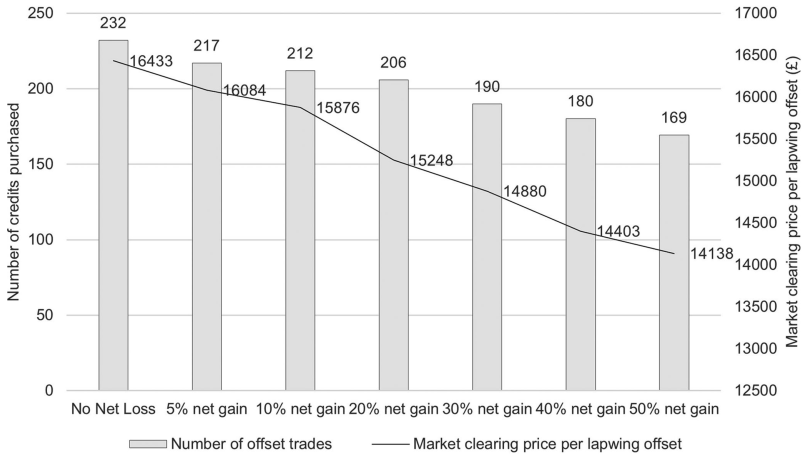

Figure 3. A comparison of the market-clearing price per lapwing offset across the net gain scenarios.

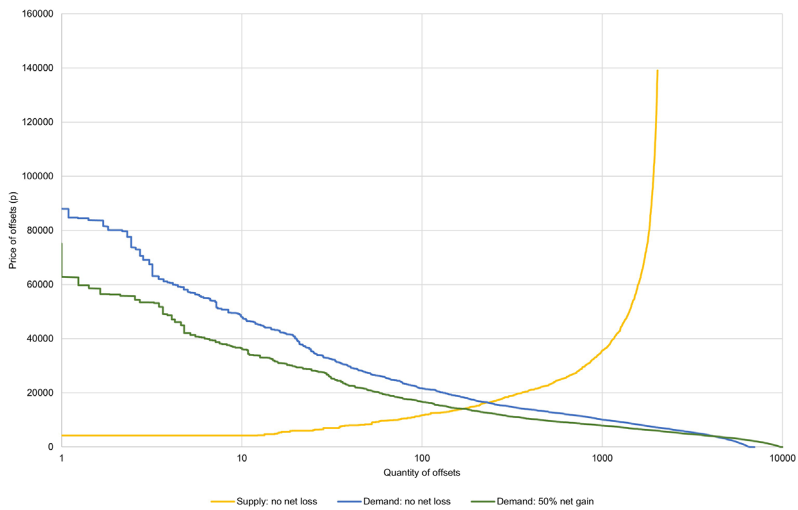

Figure 4. A comparison of offset supply and demand under no net loss and 50 per cent net gain.

Scenario 2 (no net loss), the number of land parcels developed declines to 1,042 (a reduction of 3,024 parcels) implying a cost to housebuilders of approximately £35 million in terms of revenues foregone—if we assume that the offset policy has no effect on house prices. As the net gain requirement increases from 5 per cent net gain to a 50 per cent net gain requirement, we see a continued further decline in the number of land parcels developed, from 1,018 under 5 per cent net gain down to 883 parcels developed under 50 per cent net gain. Under 50 per cent net gain, the economic cost to the housing developers in terms of foregone housing retail value is greatest, at approximately £37 million.

Finally, we can compare the ecological impacts of the no net loss and net gain policies, compared with a landscape with no offset policy in place. Under Scenario 1, where there is no offset policy in place, a total of 6,991 lapwings are lost due to development taking place on 4,066 land parcels. As expected, invoking a no net loss policy in Scenario 2 results in no losses in lapwing: the offset market clears and 232 offset trades take place between buyers and sellers of credits to allow development to take place without net losses of lapwings

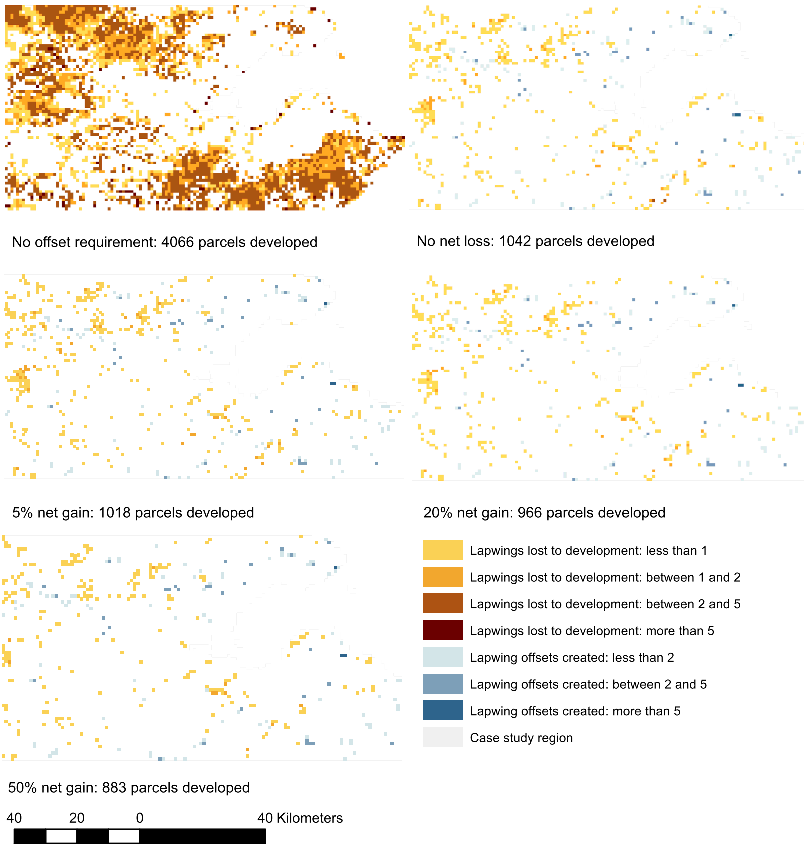

Figure 5. A comparison of development impacts across alternative net gain policy options.

across the case-study area. Moving to a net gain policy in Scenarios 3–8 has two effects on abundances of lapwing. First, the shift towards net gain results in fewer land parcels being developed as development is now less profitable, and subsequently, fewer lapwings are lost due to development impacts. Second, a move from a no net loss to a net gain policy requires developers to purchase more lapwing offsets than the number of lapwings lost due to development. For example, under the 5 per cent net gain policy target, 1,018 parcels are developed compared with 1,042 under no net loss. This results in a reduction in the number of lapwings lost from 232 under no net loss to 206 under net gain. Under the 5 per cent net gain target, developers are also required to increase the abundance of lapwing by 5 per cent, resulting in the lapwing abundance increasing by 11 (217 lapwing offsets are purchased to compensate for the 206 lost to due development) (Table 2).

We can also explore these results spatially (Fig. 5). Under Scenario 1 without development restrictions, it is clear that there would be a sustained high level of development throughout the case-study region, particularly in the northwestern and southeastern parts of the study domain. Moreover, there would be high losses of lapwing from the land parcels that would be converted in this scenario: five or more lapwings are lost on 115 parcels. Under Scenario 2 (no net loss), several predictions change. First, there is less new development. As Fig. 4 makes clear, the steep increase in the supply curve once we get beyond 235 offsets would make further development unprofitable under a no net loss policy. At the same time, there is still some new development throughout the region, particularly in the

Table 2. A comparison of the lapwings lost at the market-clearing equilibrium due to development, and lapwings generated through offset supply sites across the range of policy scenarios.

	Full development	No net loss	5% Net gain	10% Net gain	20% Net gain	30% Net gain	40% Net gain	50% Net gain
Lapwings lost at market clearing	6,991	232	206	193	172	145	129	112
Lapwings generated at market clearing	0	232	217	212	206	190	180	169
Lapwing net gain			11	19	34	45	51	57

northeastern part of the study area. Second, what development does occur avoids parcels that are particularly valuable to lapwing currently and thus would require a large number of offsets to be purchased if developed. In the majority of developed land parcels, the number of lapwings lost falls below two. Third, these losses are offset through the creation of lapwing offset supply sites, where bird populations increase. These offset supply sites are located throughout the case-study region rather than being concentrated in one specific area, although more occur north of the Firth of Forth (the major river in the maps) and closer to the coastline to the east.

Moving from Scenario 2 (no net loss) to Scenarios 3–8 (5 per cent net gain to 50 per cent net gain), more predictions change. First, we see a further reduction in development, which is in line with our previous discussions that net gain requirements make development more expensive. Development ceases on all land parcels where more than five lapwings are required to be offset (recall under 50 per cent net gain, a land parcel containing six lapwings, for example, would be required to purchase nine lapwing offsets). Under the 50 per cent net gain scenario, development only takes place on parcels requiring less than two lapwings to be offset. The concentration of development also reduces, particularly in the northwestern portion of the case-study region. Across the net gain scenarios, the number of offset supply sites remains broadly constant across the scenarios with each offset supply site choosing to supply more or fewer offsets depending on the market price of the offset credit.

5 Discussion and conclusions

Globally, there is a growing movement towards conservation policies that focus on a net gain in biodiversity (Maron *et al.* 2020). As part of these policies, developers are required by regulators to deliver a net gain in biodiversity alongside new infrastructure developments that negatively impact existing habitats and species. One option to deliver this net gain is a market for biodiversity offsets, whereby the regulator creates a market in valuable credits by imposing the offset requirement on developers, akin to a cap-and-trade pollution permit market. Tradeable offset credits offer one way in which developers can secure biodiversity net gain as part of their development plans. Our paper contributes to the emerging literature on biodiversity net gain (see Bull and Brownlie 2017; Jones *et al.* 2019; Weissgerber *et al.* 2019; Maron *et al.* 2020; Simmonds *et al.* 2020) by developing an integrated economic–ecological model that allows us to compare the economic and ecological effects of different requirements of biodiversity net gain. We compare eight different policy scenarios, ranging from Scenario 1 where there is no conservation policy in place, to Scenario 2 where there is a no net loss policy in place to Scenarios 3–8 where there is an increasing net gain requirement from 5 per cent to 50 per cent net gain. This allows us to explore how these different policies affect the demand for offset credits, in turn affecting the market price of a credit, how many trades take place between buyers and sellers of credits, and the resulting ecological landscape.

Our theoretical model predicts that the effect of increasing the net gain requirements on the equilibrium quantity (and price) of offsets is ambiguous and case dependent. That is, a biodiversity net gain policy initially increases (decreases) the equilibrium quantity and equilibrium offset price if offset demand at the no net loss equilibrium is inelastic (elastic). This is an interesting finding in itself, suggesting that the impacts of a biodiversity offset market are likely to vary according to local supply and demand conditions. Using empirical modelling for a specific case-study area, we find that increasing the net gain requirement from no net loss to a 50 per cent net gain requirement decreases both the market equilibrium price of offsets and the number of offset trades taking place. This effect is consistent throughout the net gain scenarios. As the number of lapwing offsets required by developers increases due to the changing net gain requirements from 5 per cent to 50 per cent, the developers' willingness to pay for each offset declines. This results in a decline in the offset price from £16,433 per lapwing under the no net loss scenario to £14,138 per lapwing under the 50 per cent net gain scenario. Additionally, the number of trades decreases from 232 under the no net loss policy requirement, to 217 under 5 per cent net gain down to 169 credits at 50 per cent net gain.

We thus see a continual reduction in demand for offset credits as the net gain requirements become stricter. As the number of offsets traded falls, the number of parcels on which new housing is developed also falls, from 1,042 parcels under no net less to 883 under a 50 per cent net gain target. This imposes costs on developers in terms of foregone profits from house building (assuming no effect from the offset scheme itself on house prices). The change in policy from no net loss to net gain does not shift the offset supply curve, since changing the regulatory requirement does not affect the ecological potential of land, or agricultural profits that can be obtained from this land. However, as the market-clearing price for the offset declines, fewer landowners choose to supply offsets, and thus less land is converted to conservation use.

Within our case study, there was a low level of offsets supplied at the market-clearing price across all the modelled scenarios, despite 1,400 parcels being able to offer lapwing offsets. This low level of supply mirrors some of the current problems in developing biodiversity offset markets within the UK. The first round of biodiversity offset pilot studies took place from 2012 to 2014 in the UK and a very limited number of landowners chose to engage with the pilots. Landowners voiced concerns over the costs of long-term management of offset sites, and the appropriate timescale of offset provision taking into account climate change, future development prospects for land, and cumulative development pressures in the local area (Sullivan and Hannis 2015). This lack of involvement from potential offset suppliers resulted in no offsets at all being secured in the UK pilot studies. Following the mandating of biodiversity net gain within the 25 Year Environment Plan, there is now a pressing need to re-engage with agricultural landowners who could potentially supply biodiversity offsets. In particular, incentive structures need to be revisited, with schemes needing to capture the full scale of the opportunity costs associated with converting land from agricultural to biodiversity offsets (James, Gaston, and Balmford 1999). The UK Government is currently revising UK agricultural subsidies through the new Environmental Landscape Management scheme that replaces the EU's Common Agricultural Policy. Through this, there is a shift from agricultural subsidies to payment for public goods (Bateman and Balmford 2018). One can view the use of markets in biodiversity offsets with a net gain requirement as re-enforcing this change, but here the increase in the supply of public goods (more farmland birds) is being paid for by the private sector (house builders and thus, by implication, house buyers), rather than the public sector via taxpayer funds. Whilst there are clearly overlaps between who is a taxpayer, who is a house buyer, and who owns shares in house building firms, there will be distributional impacts from a switch away from PES schemes and towards offset markets as a means of incentivising biodiversity conservation on private land.

As we would expect, the empirical results show that having any conservation policy in place, either no net loss or a level of net gain is significantly more beneficial to biodiversity than allowing unrestricted development. Indeed, a scenario of completely unrestricted development leads to a loss of over 6,000 lapwings in the case-study region. In practice, there are already planning policies in place that reduce these negative impacts, including the EU Habitats and Wildlife Birds Directives (Directives 92/43/EEC and 2009/147/EC) that designate sites as Special Protection Areas (SPAs) and Special Areas for Conservation (SACs). These are not captured within our modelling framework. Moreover, pursuing a no net loss or net gain agenda benefits other aspects of biodiversity not protected by these designations (Conway *et al.* 2013). This is especially crucial as climate change alters the range of many species and potentially moving species ranges beyond the currently protected area boundaries (Hoffmann, Irl, and Beierkuhnlein 2019).

Whilst offset policies aim to fully offset biodiversity losses due to development, there are always concerns that, depending on the design of the offset scheme, the policy involves certain ecological losses but uncertain ecological gains (Weissgerber *et al.* 2019). There will be time lags between a land parcel being developed and a supply site fully restored to a level where it is effective in providing suitable habitat to moderate the loss in biodiversity on developed sites. In such cases, there are conservation advantages from a banking approach, where credits can only be purchased once a certain level of conservation gains are realised and certified by either a government regulator or third party offset broker (Woodward and Kaiser 2002)—this is what we assume in our model. Moreover, concerns have been raised about the ecological impacts of increasing the geographic scale of an offset market, even though this increase in scale may make the market more efficient (Needham *et al.* 2020; zu Ermgassen *et al.* 2020). However, these concerns should be placed in the context of rather mixed evidence on the biodiversity benefits of current agri-environmental policies within the European Union (Batary *et al.* 2015). Finally, one can imagine that in practice farmers might be awarded offset credits, perhaps with some time lag, simply for contracting to switch to the conservation land use for the lifetime of the contract (which could be a permanent easement, but does not have to be). In such a case, the risk of the conservation practice not delivering the predicted ecological gains falls on society, whereas in the opposing case where credits are only awarded once the ecological output (more lapwings) has been observed, the risk of non-delivery falls on the farmer. This is the same policy design dilemma that we observe in the more general choice between payments for actions and payments for results (White and Hanley 2016; Zabel 2019).

We recognise that our model does not capture the intricacies associated with ecological restoration. Indeed, our model is run over a single time period, which implies that there are no time lags between the offset loss and offset gain, and that the target species can readily move between the offset sites across the region. Further, we choose to focus on one specific ecological policy target (the abundance of lapwing) and a single restoration action (the conversion of cropland to zero-grazing grassland). Our model could be extended to capture an ecological policy target that focuses on no net loss, or a net gain of specific habitats rather than a single species, or an ecological metric capturing multiple bird species with varying restoration actions. This would then allow us to examine the ecological impacts of net gain policies that allow ‘out-of-kind’ trading (zu Ermgasson *et al.* 2020). For example, with a focus on habitats, we could explore a market for environmental credits that ensure no net loss of a specific habitat, but also delivers gains in ecosystem services such as carbon storage or flood risk reduction. The ecological model used here, and the offset credit award process under consideration, also ignore any spatial agglomeration benefits from offset supply sites being located next to (or close to) each other (see Kremen and Merelender 2018). Finally, we have also not allowed for any change in house prices as a result of the operation of the offset market, although clearly localised changes in house prices could result from clusters of new developments. Based on the model set out in Section 2, this rise in local house prices would have feedback effects on the demand for offset credits, and thus on what happens to the market-clearing offset price. How big this feedback effect is depends on the elasticity of demand for offsets. For insight into the possible indirect effects of a biodiversity offset market on non-target ecological indicators, and results on the implications of changing the geographic scale of the offset market, see Needham *et al.* (2020).

We also acknowledge that our model applies in a world of perfect information, in the sense that the demand curve for credits reflects the actual value of land for housing (so that this value is not private to each buyer), whilst landowners do not succeed in obtaining information rents in addition to the opportunity costs of switching to conservation land use, given that these opportunity costs are only known fully by the farmer in reality. Offset markets rely on competition between multiple offset suppliers to keep information rents low (indeed, to keep them at zero in our framework), rather like conservation auctions relying on competition between competing offers to minimise information rents. Moreover, if we recognise that switching to a permanent conservation land use might be irreversible from the landowner’s perspective—assuming that such a contract can be constructed and enforced—then suppliers might demand a payment higher than the present value of all opportunity costs, to compensate them for this loss of option.

As mentioned above, markets in biodiversity offsets have some similarities as an economic instrument with Tradeable Pollution Permits (TPPs), since the regulator creates a valuable commodity by imposing a ceiling or cap on economic activity. In the offset market, this establishes a price which, in principle, could lead to the cost-effective allocation of land to conservation versus development, just as the permit price in a TPP market provides a signal to encourage least-cost abatement of pollution. A key difference, however, is in the supply side of these created markets. In a TPP market, the total supply of permits is determined by the regulator. Individual firms decide how many permits to buy and/or sell at a price that equates this exogenous aggregate supply to the aggregate demand curve for permits, which in turn depends on marginal abatement costs. In a market for offsets, in contrast, aggregate supply is endogenous: individual farmers compare the opportunity costs of agricultural production with the willingness to pay of housebuilders to determine how many offsets to create and then supply, although the regulator established the maximum permissible overall impact of economic activity on the environmental outcome in focus (here, birds). However, an interesting and largely unexplored research question is what lessons we can learn from TPP markets that apply to markets in biodiversity offsets (Needham *et al.* 2019).

Finally, whilst we have employed data from a specific UK case study, our analysis provides an approach that is generalisable across countries looking to expand net gain policies and biodiversity offset markets. It would be interesting to replicate this approach across varying ecological, agricultural, and development gradients, and to figure out what matters most in determining the direction and extent of changes in economic and ecological outcomes of an offset market as the net gain target is increased.

Supplementary material

Supplementary data are available at *Q Open* online.

Acknowledgement

We thank the editor and two anonymous referees for insightful comments on an earlier version of this paper.

Funding

We thank the Leverhulme Trust and the European Commission (under the EFFECT project) for part-funding this work.

Data Availability

The data sources used in working on this paper are described in full in the Supplementary Material.

End Notes

1. The move towards net gain sits alongside a broader shift in UK agricultural policy. A new Environmental Land Management scheme will replace schemes available to farmers under the EU's Common Agricultural Policy, as Britain leaves the EU. The Environmental Landscape Management scheme will reward farmers and land managers with public money for the provision of public goods, including improvements in biodiversity, cleaner air and water, healthier soils, and natural hazard protection (Defra 2020).
2. Needham *et al.* (2020) look at a no net loss target for three different ecological indicators.
3. Needham *et al.* (2020) look at no net loss in the Tees Estuary, England.

References

Barker Nicole K. S. et al. (2014) 'Modeling distribution and abundance of multiple species: different pooling strategies produce similar results', *Ecosphere* 5: art158. https://doi.org/10.1890/ES14-00256.1.

Batary P et al. (2015) 'The role of agri-environmental schemes in conservation', *Conservation Biology,* 29: 1006–116.

Bateman I.J. and Balmford B. (2018) 'Public funding for public goods: a post-Brexit perspective on principles for agricultural policy', *Land Use Policy*, 79: 293–300.

BBOP (Business and Biodiversity Offsets Programme). (2009) Biodiversity offset design handbook.

Beattie A. (2019) 'The Farm Management Handbook'. https://www.fas.scot/downloads/farm-management-handbook-2019-20/. Accessed 9 March 2021.

Bull J.W. and Brownlie S. (2017) 'The transition from no net loss to a net gain of biodiversity is far from trivial', *Oryx*, 51: 53–9.

Bull J.W. and Strange N. (2018) 'The global extent of biodiversity offset implementation under no net loss policies', *Nature Sustainability*, 1: 790–8.

CIEEM C., IEMA. (2016) *Biodiversity net gain: Good practice principles for development*.

Conway M. et al. (2013) 'Exploring potential demand for and supply of habitat banking in the EU and appropriate design elements for a habitat banking scheme'. In: E. Commission (ed.), ICF GHK, Bio Intelligence Service, London.

Dinerstein E. et al. (2020) 'A "Global Safety Net" to reverse biodiversity loss and stabilize Earth's climate. Science advances', 6(36): eabb2824.

Eaton M.A. et al. (2015) 'Birds of Conservation Concern 4: the population status of birds in the UK, Channel Islands and Isle of Man', *British Birds*, 108: 708–46.

Gibbons P. and Lindenmayer D.B. (2007) 'Offsets for land clearing: no net loss or the tail wagging the dog?', *Ecological Management & Restoration*, 8: 26–31.

HM Government U.K. (2018) '25 Year environment plan'. www.gov.uk/government/publications. Accessed 1 December 2020.

Hoffmann S., Irl S.D. and Beierkuhnlein C. (2019) 'Predicted climate shifts within terrestrial protected areas worldwide', *Nature communications*, 10: 1–10.

IPBES (2019) 'Summary for policymakers of the global assessment report on biodiversity and ecosystem services'. In: S. Díaz et al. IPBES secretariat, Bonn, Germany. https://doi.org/10.5281/zenodo.3553579

James A.N., Gaston K.J. and Balmford A. (1999) 'Balancing the Earth's accounts', *Nature*, 401: 323–4.

Jones J. P. G. et al. (2019) 'Net gain: seeking better outcomes for local people when mitigating biodiversity loss from development', *One Earth*, 1: 195–201.

Kremen C. and Merenlender A.M. (2018) 'Landscapes that work for biodiversity and for people', *Science* 362: eaau6020.

Laitila J., Moilanen A. and Pouzols F.M. (2014) 'A method for calculating minimum biodiversity offset multipliers accounting for time discounting, additionality and permanence', *Methods in Ecology and Evolution*, 5: 1247–54.

Maron M. et al. (2018) 'The many meanings of no net loss in environmental policy', *Nature Sustainability*, 1: 19–27.

Maron M. et al. (2020) 'Global no net loss of natural ecosystems', *Nature Ecology & Evolution*, 4: 46–9.

Nature. (2020) 'The United Nations must get its new biodiversity targets right', *Nature*, 578: 337–8.

Needham K. et al. (2019) 'Designing markets for biodiversity offsets: lessons from tradable pollution permits', *Journal of Applied Ecology*, 56: 1429–35.

Needham K. et al. (2020) 'Understanding the performance of biodiversity offset markets: evidence from an integrated ecological–economic model', *Land Economics*. In press.

Rayment M. (2019) 'Paying for public goods from land management: how much will it cost and how might we pay. rayment consulting services ltd. A report for the RSPB, the national trust and the wildlife trusts DEFRA'. 2020. https://www.gov.uk/government/publications/the-environmental-land-management-scheme-an-overview. Accessed 9 March 2021.

Rowland C.S. et al. (2015) 'Land Cover Map 2015 (vector, GB)'. NERC Environmental Information Data Centre. https://doi.org/10.5285/6c6c9203-7333-4d96-88ab-78925e7a4e73.

Scottish Government. (2020) 'The environment strategy for Scotland: vision and outcomes'. https://www.gov.scot/publications/environment-strategy-scotland-vision-outcomes/. Accessed 23 May 2020.

Scottish Natural Heritage. (2019) 'Scottish biodiversity strategy'. https://www.nature.scot/scotlands-biodiversity/scottish-biodiversity-strategy. Accessed 23 May 2020.

Simmonds J.S. et al. (2020) 'Moving from biodiversity offsets to a target-based approach for ecological compensation', *Conservation Letters*, 13: e12695.

Sullivan S. and Hannis M. (2015) 'Nets and frames, losses and gains: value struggles in engagements with biodiversity offsetting policy in England', *Ecosystem Services*, 15: 162–73.

United Nations. (2018) *The sustainable development goals*. UN.

Weissgerber M. et al. (2019) 'Biodiversity offsetting: certainty of the net loss but uncertainty of the net gain', *Biological Conservation*, 237: 200–8.

White B. and Hanley N.D. (2016) 'Should we pay for ecosystem service outputs, inputs or both?', *Environmental and Resource Economics*, 63: 765–87.

Woodward R.T. and Kaiser R.A. (2002) 'Market structures for US water quality trading', *Applied Economic Perspectives and Policy*, 24: 366–83.

Zabel A. (2019) 'Improving private land conservation with outcome-based biodiversity payments', *Journal of Applied Ecology* 55: 1476–85.

Zu Ermgassen S. et al. (2020) 'The hidden biodiversity risks of increasing flexibility in biodiversity offset trades', *Biological Conservation*, 252, 108861.