Understanding the Performance of Biodiversity Offset Markets: Evidence from An Integrated Ecological – Economic Model

Dr Katherine Hannah Simpson

Affiliation: Institute of Biodiversity, Animal Health & Comparative Medicine, University of Glasgow, Glasgow, UK

Mailing Address: Graham Kerr Building University of Glasgow, 82 Hillhead Street, Glasgow, G12 8QQ

Katherine.simpson@glasgow.ac.uk

Professor Frans de Vries

Affiliation: Department of Economics, University of Aberdeen Business School

Mailing address: Department of Economics, University of Aberdeen Business School, Aberdeen AB24 3QY, Scotland, UK

E-mail: frans.devries@abdn.ac.uk

Dr Martin Dallimer

Sustainability Research Institute, School of Earth and Environment, University of Leeds, Leeds, UK

Mailing address: School of Earth and Environment, University of Leeds, Woodhouse Lane, Leeds, LS2 9JT

M.Dallimer@leeds.ac.uk

Professor Paul R. Armsworth

Department of Ecology and Evolutionary Biology, University of Tennessee, Knoxville, Tennessee

Mailing Address: 569 Dabney Hall, 1416 Circle Drive, Knoxville, TN 37996-1610

p.armsworth@utk.edu

Professor Nick Hanley

Institute of Biodiversity, Animal Health & Comparative Medicine, University of Glasgow, Glasgow, UK

nicholas.hanley@glasgow.ac.uk

Abstract

Biodiversity offset markets can incentivize private landowners to take actions that benefit biodiversity. A spatially explicit integrated ecological-economic model is developed and employed for a UK region where offset buyers (house developers) and sellers (farmers) interact through trading offset credits. We simulate how changes in the ecological metric and geographic scale affects the performance of the offset market. Results show that the choice of the metric has a significant effect on market liquidity and the spatial distribution of gains and losses in the “target” species. The results also consistently reveal relatively higher potential welfare gains for developers than for farmers.

#  Introduction

Biodiversity offsetting aims to reconcile the mounting pressure for urban development and agricultural expansion with the need to significantly reduce the impacts of these on biodiversity (Ten, Treweek, and Ekstrom 2010). Biodiversity offsets provide measurable conservation gains to compensate for significant, residual impacts on biodiversity as a result of new development activities (BBOP 2009). The term “offsetting” often encompasses a wide range of mechanisms including compensatory mitigation, in-kind compensation, mitigation banking, habitat banking, species banking, and wetland banking (Lapeyre, Froger, and Hrabanski 2015).

In this paper, we are specifically interested in *markets* for biodiversity offsets. These markets are created when multiple buyers and sellers of offset credits interact with others through a trading process. This creates a setting in which 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 on some measure of biodiversity. Such trades can be facilitated by an offset bank which collects offers from sellers and makes these available to potential buyers. By establishing an appropriate rate of exchange between sellers and buyers, markets can, in theory, achieve no net loss of biodiversity within some defined area at least cost.

Through the creation of economic incentives for conservation, market mechanisms such as markets for offsets encourage private landowners and firms to take costly actions that benefit biodiversity (Kangas and Ollikainen 2019). A biodiversity offset mechanism can steer land allocation choices in the direction of greater efficiency, given some constraint on overall biodiversity change. 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 of creating offsets, through foregoing some other form of land use such as crop production, or through ecological restoration activities. Offset markets allow biodiversity conservation targets to be met at the lowest cost since only those suppliers whose supply prices are competitive will be rewarded with payments for supplying offsets. Moreover, the establishment of a market-facilitation bank allows developers to purchase “off the shelf” offsets (Bonds and Pompe 2003; Levrel, Scemama, and Vaissière 2017; Hannis and Sullivan 2012). Offset banks allow the purchase of credits where ecological gains have already been certified, overcoming one of the widely recognised problems in offsetting, namely that the ecological benefits of restoration actions are uncertain (Maron et al, 2012).

Despite an increasing policy interest in biodiversity offsets and a growing ecological literature, relatively little attention has been paid to the idea by environmental and resource economists when compared to other incentive-based mechanisms, such as payments for ecosystem services and tradable pollution permit markets for air and water pollution (Coralie, Guillaume, and Claude 2015; de Vries and Hanley 2016). From an economist's perspective, creating markets for biodiversity offsets faces policy designers with some of the same issues as with tradable pollution permit markets, including what to trade (credit definition), market participation and geographic scale, and the establishment of appropriate trading ratios (Bonds and Pompe, 2003; Wissel and Wätzold, 2010; Needham et al, 2019). Our paper contributes to this literature by exploring two of these design issues, namely credit definition and market scale, showing how policy design choices affect the economic and ecological outcomes of offset trading. Specifically, we aim to answer the following questions:

1. How does the choice of what to trade affect the ecological and the economic outcomes of an offset market?
2. How does the geographic scale of the market affect these outcomes?

To answer these questions, we first develop a conceptual model of a biodiversity offset market in the context of a no net loss of biodiversity regulation. Spatially-explicit biodiversity offset supply and demand curves are then constructed using integrated ecological-economic modelling, in an application of this conceptual framework. These supply and demand curves capture the spatial variations in the costs of supplying biodiversity offsets. This spatial variation depends on the relative value of land for agriculture and the value of new housing developments across the landscape. Through the identification of the market-clearing price for the biodiversity offset, our model can then be used to assess the economic welfare implications of such a market in terms of the distribution of benefits between sellers and buyers. Importantly, we are also able to estimate the ecological implications of such a market across the landscape to identify where the gains and losses in biodiversity take place and to which species. Our empirical model captures (i) how the choice of what to trade has significant implications on the scale and distribution of new development, (ii) how the choice what to trade has significant impacts on ecological gains and losses of species not protected under the no net loss regulation and (iii) the implications of changing the geographic scale of the market.

The remainder of the paper is organised as follows. In the next section, we discuss design issues in offset markets and the related economic literature. In Section 3 we develop our conceptual model of biodiversity offset trading. Section 4 details the empirical approach, including an overview of the integrated economic - ecological model, our case study area, data sources and data development (much more detail can be found in the SI accompanying this paper). Results are presented in Section 5 and the discussion and conclusion in Sections 6 and 7.

# Background

## Designing an offset market

Needham et al, (2019) review experience from pollution permit markets and identify some fundamental parameters that seem relevant for the design of biodiversity offset markets. The most important of these from the perspective of the current paper are the following.

1. What to trade: policy targets and ecological metrics

When establishing a pollution permit-trading scheme, the first stage is for the regulatory agency to determine the cap on overall emissions. While voluntary trading can occur (for example, in the context of carbon offsets and forest planting), imposing a regulatory cap greatly increases the volume of trading. Efficient environmental markets require goods to be grouped into simple, measurable, standardised units in order so that they are fully exchangeable. For biodiversity offsetting, a vague cap dictating “no net loss of biodiversity” leaves room for ambiguity and is difficult to measure and quantify (Bull et al, 2013). This undermines effective monitoring and enforcement, with the prospect of poor market functioning in terms of lack of both cost-effectiveness and environmental effectiveness. A clear *metric* for how offsets are measured is thus required. What exactly this metric should be is much less obvious. Our paper compares three candidate metrics, to examine the effects of this choice on market performance.

1. The trading ratio

This governs the rate of exchange between offsets at different points in space (between the supply site and the demand site for any prospective trade). Once the metric for an offset market has been established, trading ratios are then required to determine the ecological rate of exchange between sellers and buyers at different points in space (Bull et al, 2017). Ideally, such trading ratios need to reflect the many potential sources of ecological heterogeneity implicit in the spatial definition of the credit, and thus the ecological consequences of moving pressures on the biodiversity metric (positive, for offset supply and negative, for offset demand) across different locations within the policy area (Maron et al, 2012; McKenney and Kisecker, 2010). The parallel concern in polluting trading has been to try to set trading ratios to prevent local breaches of ambient quality targets even when aggregate emissions fall within the cap (Atkinson and Tietenberg, 1987). The ecological literature has also discussed the role of trading ratios to reflect uncertainty over the ecological benefits generated by actions taken to create the offset, and time lags in how long these ecological benefits take to emerge (Bull et al, 2013). However, in this paper we restrict the role of trading ratios to reflect ecological heterogeneity over space alone.

1. Market scale and trading volume

Biodiversity offset trading takes place within a regulated geographic area. Setting these spatial restrictions suggest the possibility of economic and ecological trade-offs (see Womble and Doyle, 2012; Doyle et al, 2014). There is an implication that larger geographic service areas would be preferred to smaller service areas on grounds of economic efficiency, since larger service areas imply more potential agents to trade. In contrast, smaller service areas imply fewer opportunities to trade. In the latter case, thin markets can lead to erratic price signals and may increase the uncertainty on investor returns from development and, in turn, increase development costs (Fisher-Vanden and Olmstead, 2013). Higher offset price volatility may also deter potential suppliers from incurring the (opportunity) costs needed to create credits. However, larger spatial markets imply a greater heterogeneity in the biodiversity value of conservation actions at different sites, making it harder to measure and ensure equivalence in ecological gains and losses across space (BenDor and Brozović, 2007).

In the design of biodiversity offset markets, there will likely be a case-specific optimal trade-off between having a larger spatial scale which increases participation, and a smaller spatial scale which makes the determination of ecological equivalence easier to model and control for via trading ratios. Moreover, ensuring transactions costs of trading are kept as low as possible seems likely to be important. In our paper, we thus explore the effects of changing the size of the market on economic and ecological performance.

Fernandez and Karp (1998) provided one of the first economic analyses of wetland banking, focussing on the optimal level of investment in wetland mitigation banking in the USA. Using an ecological-economic model they considered how restoration costs, restoration potential and the market value of offset credits determine how much a landowner should invest in a wetlands bank. Following ecological expectations, the authors found that as restoration costs reduce, it was more likely a landowner would restore his whole land parcel. Furthermore, increasing ecological uncertainty leads to higher credit prices. Heal (2005) compared a habitat-hectares offset approach with species banking to protect the red-cockaded woodpecker, and found that more restrictive definitions of the ecological metric increase the price of a credit. Furthermore, choosing the incorrect ecological metric nullifies the ecological benefits of the trading system.

Other modelling approaches have focused on the impact of time lags in restoring habitats and the subsequent impact on species. Time lags have been shown to increase the restoration costs for biodiversity offsets and lead to greater fluctuations in credit supply prices (Drechsler and Hartig 2011). Solutions to include developing a “credit release schedule” to foster investment into offset sites. The credit release schedule needs to balance releasing enough “early release credits” (credits which have not yet reached the full restoration potential) and not over releasing credits which leads to a future net-loss in habitats and species (BenDor, Guo, and Yates 2014).

Implications of the trading ratio on market functioning have also been a key theme within the economics literature. Bonds and Pompe (2003) provide one of the first economic analyses for calculating a trading ratio that adjusts for the functionality and location of wetlands. Further work has been undertaken in the ecological literature by BenDor et al, (2009); Bruggeman et al, (2005); Bull et al, (2014); Laitila et al, (2014); Moilanen et al, (2009). Further research has concentrated on linking economic and ecological models to examine whether offset markets offer new opportunities for biodiversity conservation. Doyle and Yates (2010) provide an economic-ecological model of ecosystem service markets following a no net loss principle. A focal point in their analysis is the scale of the market by assessing how (free) entry of mitigators (i.e., suppliers of offset credits) affects ecosystem restoration. Their results indicate that, as additional suppliers enter the market, the size of an individual restoration project decreases but the total amount of mitigation increases, thus increasing credit supply. Their results also indicate that using a single metric to capture ecosystem functioning will result in loss of some ecosystem functions, stressing the need to further understand the relationship between restoration and multiple ecosystem service functions.

Most recently, Kangas and Ollikainen (2019) develop a model to analyse a market for biodiversity offsets in Finland with a focus on the restoration potential of three habitats: wetlands, forests and agricultural habitats. Credits are developed as a result of restoration measures which would not have otherwise occurred, and restoration potential is assessed based on expert assessments and the literature. Demand is based on predictions for future land use change. The model considers how market equilibrium adjusts in response to trading ratios, the focus habitat for restoration, and uncertainties associated with restoration outcomes. Results showed that all three of these aspects impacted the market equilibrium price in theoretically anticipated ways: offset prices were higher when restoration actions were more costly and uncertainty high.

Building from the above literature, our research uses an integrated ecological-economic modelling approach to explore first theoretically, and then empirically (for a specified UK case study), the impact of two design parameters on equilibrium price for biodiversity offset credits. We contrast economic and ecological market outcomes under varying species-specific ecological policy targets and different geographical trading regions. In addition, our model allows us to assess how land-use and specific species distribution and abundance shift within the case study region as a result of offset trading.

# A Conceptual Model of Biodiversity Offset Trading

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 conservation. Within such a setting, it is assumed that there are a number of farmers who own the land and developers who wish to acquire land for housing development. We further assume that both the 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.

Farmers’ default land use is assumed to be for agricultural purposes in the form of crop or livestock production. However, farmers can offer up some of their land for conservation, where conservation land (newly created wildlife habitat, for instance) earns offset credits depending on the biodiversity metric used. The farmer’s optimization problem entails maximizing the profits derived from agricultural production plus the value of any offset sales. Every hectare given up to create new habitat means one less hectare for agricultural production. Furthermore, conversion to a different habitat type may involve a restoration cost. Therefore, the conversion cost to the farmer consists of the opportunity costs of the foregone agricultural output plus any associated restoration costs.

From an agricultural perspective, land is of variable quality, measured in terms of agricultural output per hectare. Ranking all land owned by any one farmer along this gradient yields a continuous, upward sloping supply (marginal cost) curve for creating new offset credits. The farmer maximises profits by choosing to supply the quantity of offsets that equates the marginal cost of creating new habitat (i.e., lost agricultural profit at the margin) with the offset price. If agricultural prices rise, then the opportunity cost of creating new habitat rises, and so the supply curve for each individual farmer (hence aggregate supply) would shift up.

House developers face land values for housing that varies spatially across the region. Each developer knows that, depending on the conservation value of land, each hectare acquired for new housing development requires a number of offset credits to be purchased from an offset provider or offset bank. Ranking development options according to their expected housing value to this developer from highest to lowest yields a downward sloping demand curve for offset credits. The developer chooses to buy a quantity of credits which equates their individual demand curve with the market price of offsets[[1]](#endnote-1). If the expected average house price in the region rises, this would increase the developer’s willingness to pay for any potential housing site, implying a shift of the demand curve for offsets to the right. The slope of the aggregate demand curve depends on the heterogeneity in the derived demand for offsets, i.e., developers’ willingness to pay for land for housing development. If each hectare of habitat created can be heterogeneous in ecological terms, then this means that each offset created or demanded has a different ecological value. On the demand side, higher ecological value land requires credits of higher value to be purchased to offset the impacts of development on biodiversity; on the supply side, higher value ecological land generates more valuable credits when new habitat is created.

Assume for the moment that we have a single ecological indicator which is the focus for policy. Then we can think of the value of any given credit supplied to the market to be , with referring to the amount of hectares and indicating the average ecological value at site . On the demand side, a developer must obtain a number of offsets equal to a value , where reflects the ecological value of site which is damaged by the development. Assuming a development of one hectare in size, the developer must ensure it purchases appropriate offsets such that . From this, we can define the offset trading ratio between any two sites (supply side) and (demand side) as:[[2]](#endnote-2)

 [1]

We are now in a position to develop a simple model that highlights the core functioning of the market for biodiversity offsets. This not only allows us to draw some clear comparative statics results, but also provides a solid basis for the actual case study implementation later. We will do so by building on Doyle and Yates (2010) but extend their model in two important ways. First, our model captures heterogeneous opportunity cost of land use. Second, given this heterogeneity, we will derive spatially explicit demand and supply functions for biodiversity offsets.

Given a number of farmers, the quantity of hectares (offsets) “produced” by an individual seller can be represented by the function , where is the offset price, is the trading ratio as defined in [1], and the opportunity cost of land use of type . Aggregating across all farmers, the total amount of land supplied for conservation, , is then:

 [2]

On the demand side there are a total of developers. These developers must buy a number of hectares to offset the degradation of each hectare due to development. Analogous to the supply side of the market, given the trading ratio [1], the number of developers (), and the offset price (), the demand function of an individual developer is , where is the value of land for housing in the development area. The total amount of hectares demandedfor development, , follows a straightforward aggregation across all developers:

 [3]

The aggregate demand curve for offset credits in [3], representing a particular set of spatial values for the value of land for housing development, and the aggregate supply curve for offsets in [2], showing the value of foregone profits from farming, allows the offset market to clear at an equilibrium offset price . This generates an equilibrium amount of hectares for biodiversity offsets such that Knowing the expression of the equilibrium price of offset credits, , we can now do some comparative statics exercises to determine how the equilibrium price responds to changes in the identified determinants. First, a change in the market size governs the offset equilibrium price in a straightforward manner. An increase in the number of sellers, , would increase the amount of hectares supplied to the market for offsetting, which would have a downward pressure on the equilibrium price. In contrast, the equilibrium price would increase if demand would be enhanced through an increase in the number of buyers, . These two effects are summarized by Eqs. [4a] and [4b], respectively:

 [4a]

 [4b]

Second, a rise in the opportunity cost of land use of type , implying more profit foregone from agricultural production, has a suppressing impact on the total amount of land supplied at any price. This, in turn, would push up the equilibrium price of offsets for a given demand for offsets:
 [4c]

Third, higher expected house prices would attract more land being demanded for development. For a given supply of land, this would increase the equilibrium offset price:

 [4d]

Finally, a change in the trading ratio between site and can originate from a change in the ecological value of either site. More specifically, following the definition of the trading ratio in [1], an increasing trading ratio can be the result from a higher (lower) ecological value of site (). If the ecological value of site increases, this reduces the number of offset credits a developer needs to buy, since each credit now delivers more biodiversity. The same effect applies if the ecological value of site decreases. In this case too the developer needs to buy less credits to offset each hectare of land on which development occurs, since the land converted to development is ecologically less valuable. This reduction in offset credits demanded implies that the equilibrium offset price decreases with an increasing trading ratio:[[3]](#endnote-3)

 [4e]

# An Integrated Ecological – Economic Model of a Biodiversity Offset Market

# Model Overview

In the previous section, we developed a conceptual model for a biodiversity offset market. We now adapt this model and apply it to our case study, using integrated ecological-economic modelling. This empirical model seeks to maximise the joint agricultural and development profits of landowners across a defined region, subject to a regulatory limit of no net loss of biodiversity.

Each landowner manages a single 1 by 1 km land parcel and has three options: develop land for housing, provide biodiversity offsets, or maintain the current land use. We assume that landowners maximise profits from land use for each parcel. This profit maximization is subject to a number of constraints: land that is already developed for housing can only remain in a developed state; agricultural land can either remain as agricultural land, or the farmer can undertake actions which benefit the ecological target and thus generate biodiversity offset credits. If a landowner chooses to develop, he must hold the relevant number of offset credits to ensure no net loss of the ecological target in the land parcel.

Our empirical approach consists of three stages. Firstly, an ecological model predicts the current level of our ecological target variable across the case study region, based on current land use. This provides us with a no net loss baseline for the ecological target. Secondly, the ecological model is used to estimate *potential* changes in the ecological target as a result of landowners undertaking actions which benefit the ecological target, thus generating the potential number of biodiversity offset credits a land parcel could supply. We then determine the profitability of each land parcel under each land-use options, i.e., development, offset provision or current land use. By integrating this profitability with the offset requirements, supply and demand for offset credits for each land parcel is determined. Finally, we model optimal, sequential trades based on derived spatially derived demand and supply curves and the no net loss policy goal. We assume that a mechanism exists within the offsets market which (i) collects supply offers from all potential suppliers (farmers), in terms of their minimum willingness to accept compensation for the offer of a given offset credit; (ii) collects demand offers from all potential buyers, in terms of their maximum willingness to pay for each offset credit; (iii) orders these supply and demand offers from highest to lowest (demand) and lowest to highest (supply); then (iv) pairs potential buyers and sellers sequentially in order of (highest WTP / lowest WTA) to (lowest WTP / highest WTA) until no more gains from trade can be realised. This can be called a Walrasian market-clearing mechanism through a hypothetical auctioneer[[4]](#endnote-4). A schematic of the modelling process is provided in Figure 1.

**[Insert Figure 1 here]**

## UK Case Study

We apply our model of biodiversity offsetting to a UK case study region known as the Tees Valley, Pennine Uplands and North York Moors (Figure 2) The case study region covers an area of approximately 5400 km2 and encompasses a range of habitats and land use types, from upland moors in the west of the region, low lying agricultural land throughout the central region and increasing urbanization in the east at the coastal margin.

The case study region includes three Special Protection Areas (SPAs) designated under the EU Birds Directive and three Special Areas for Conservation (SACs) designated under the EU Habitats Directive. The upland habitats of the North Pennine Moors SPA and SAC and North Yorkshire SPA contain extensive tracts of semi-natural moorland habitat including upland heath (heather and blanket bogs) and calcareous grasslands. The case study region falls within the Northern Upland Chain Local Nature Partnership that contains the highest density of breeding curlew in Europe and has thus become a priority for stabilising the breeding population (Northern Upland Chain Local Nature Partnership, 2020).

The coastal habitats of the Teesmouth and Cleveland Coast SPA are so classified for the assemblage of over 20,000 wintering warders, including curlews, lapwings and oystercatchers (JNCC, 2020). The area has been highly modified by human activities, with over 90% of intertidal habitats lost to land claim (Smith,, 2011). Local development plans have highlighted the need to maximise the river and port as key economic assets (Stockton-On-Tees Borough Council, 2016). Further expansion of the port, and the chemical and process industries which is supports are expected to place further pressures on these coastal habitats (Simpson, 2011). Within the wider Tees Valley, an increase in 27,000 households is projected by 2039 requiring an increase in the affordable housing stock of 1,832 to 2,106 dwellings per annum (Ferrari and Dalgleish, 2019).

At the national scale, the UK Planning Regulations are currently being reviewed as part of the 25 Year Environment Plan (HM Government, 2018). The UK Government has committed to building 300,000 new homes per year up to 2025 whilst safeguarding biodiversity. This includes embedding a “Biodiversity Net Gain” principle for development and creating or restoring 500,000 hectares of habitat. Specifically, it identifies that this net gain could be delivered by “habitat banks, land-owners or brokers as part of a flexible market” (HM Government, 2018). This decision of the UK Government to pursue offset markets, and the competing pressures for development and conservation of biodiversity in the Tees region, make this an ideal study region in which to test our general model of biodiversity offsets.

**[Insert Figure 2 here]**

## Market Design Parameters

#### *Policy target and ecological metric*

As previously highlighted, the first stage of developing a biodiversity offset market should be the identification of the policy target to determine the ecological metric which denominates an offset credit. Based on the international importance of the case study region for supporting breeding and overwintering bird populations, the policy target for our modelling approach is no net loss of each of three wading bird species: the Eurasian oystercatcher (*Haematopus ostralegus*), Eurasian curlew (*Numenius arquata*) and the northern lapwing (*Vanellus vanellus*). These species were chosen based on national and regional priorities for wader conservation: all three species are recorded as ‘Near Threatened’ on the international IUCN Red List4 of Threatened Species (Bird Life International, 2019). Of particular concern is the decline in curlew populations, with UK populations declining at the steepest rate recorded throughout the curlew’s range in Europe. Population declines are linked to a loss and degradation of breeding habitat as a result of changing agricultural practices, such as the conversion of traditional meadows to intensive grassland monocultures (Franks et al, 2017).

One model of biodiversity offsetting will be constructed for each species separately, that is each model focusses on no net loss of a single target species. This follows the recommendations of Needham et al, (2019) which require that the biodiversity offset market to be based on a simple, measurable ecological metric which can be denominated in standardised units. This approach allows us to ensure no net loss of the target species, but to also consider what we call the “unintended consequences of trading”: what are the ecological impacts on the two species not considered under the no net loss policy target?

In addition to our policy target, the regulator also sets a “preferred land management practice” for generating offset credits. The land management practice is one which is *expected* to benefit the target species. By replacing the current land use with this new, ecologically preferred land management practice, the expected abundance of the target species will increase, hence generating biodiversity offset credits. For our model, we adopt a preferred land management practice of “improved grassland” without livestock. This change in land management practice will only take place on agricultural land patches currently farmed for crops or livestock, as such we make the assumption, based on modelling species abundances across the region (see Supplementary Information) that improved grassland is more beneficial to the three target species than the agricultural practices of crop and/or livestock production.

It is expected that different land management practices will have different costs associated with them. Restoration costs associated with grassland conversion from agricultural land currently used for cropping are minimal, typically involving soil cultivation and seeding only. However, under a different preferred land management practice, such as wetland restoration, the restoration costs would be expected to be significantly greater.

#### *Market scale*

We compare the ecological and economic efficiency of an offset market at different spatial scales for the oystercatcher policy target. The first analysis allows trading across the whole of the case study region i.e. there are no restrictions on which parcels can trade. The second analysis places geographic restrictions on which parcels can trade. The case study region is divided into three geographically smaller service areas (known as planning areas) and trades are restricted so that trades can only take place within, but not between, each of these areas.

## The Empirical Modelling Framework for the Offset Market

As a baseline, we take the current land use structure in the case study area of size . This is divided into a number of land parcels equalling a size of 1 x1 km. There is a single agent (farmer) controlling each land parcel. Each land parcel can comprise of any combination of 30 distinguished land use types and crop classifications, i.e., (Rowland et al, 2015).

A policy target is chosen based on no net loss in the abundance of a single wading bird species which thus forms the ecological metric. Each land parcel has an associated ecological index , which is the baseline (current) abundance of the single species found within the corresponding land parcel. Across the whole case study area, total species abundance is then simply the aggregate , which represents the no net loss conservation objective.

The regulator sets the preferred land management practice that benefits the target species and this allows us to identify the land parcels that offer biodiversity offsets. Offsets are generated based on a farmer switching the entire land parcel to the new land management practice. This provides us with a measure of single species abundance for each land parcel before and after the change in land management practice. This subsequently allows us to calculate the gains and losses for the species within the land parcel. Recall that we assume that the default crop and livestock distribution on agricultural land is currently the most profitable to a farmer, and that switching to an alternative land management practice will (potentially) result in a loss of profit. Consequently, for a farmer to choose to supply offsets, the offset price must at a minimum compensate for this loss in profit. We can then calculate the farmer’s minimum willingness to accept for a change in land management practice, which gives us the minimum unit price farmer would be willing to supply one offset:

 [5]

Here denotes the increase in the target species () abundance gained from offsetting the parcel (again, a binary decision to shift all cropland to improved grassland). Therefore, with being the market offset price, reflects the unit price of a single wading bird species gained from the offset. This allows us to generate a series of supply prices for a single offset for each land parcel. To do so, given the distinguished land use and crop classifications, we first calculate the farmers’ profit by using crop and livestock gross margins (revenues minus variable costs) data within each land parcel :

 [6]

Where is the total land area within the parcel on which a specific agricultural output occurs, is the gross margin for land use type in period , with and referring to the corresponding values *before* and *after* changing the agricultural land management practice, respectively.

For a farmer to enter the market as an offset supplier he needs to be compensated for the difference in values between his land under crop and livestock production and the value under offset production[[5]](#endnote-5):

 [7]

Knowing the opportunity cost that follows from [7] and taking the predicted increase in the abundance of the target species from the ecological model, land parcels can now be ranked based on which deliver the “best” value biodiversity offsets. Substituting these values into [5] gives a straightforward basis on which to rank of parcels, where the inverse of [5] is essentially a benefit/cost ratio of gains in the target species.

On the demand side of the market, developers’ demand for offset credits will be determined by the expected value of land for housing development, , taking into account the need to purchase credits to offset any losses in the target species:

 , [8]

with referring to the maximum willingness to pay for an offset by developer .

We assume that all 1 km land parcels can be developed within our model regardless of whether a parcel already contains some level of development or not. However, there will be hectares within the land parcel which cannot be developed on, either as the land already contains development (classified as suburban and urban in the habitat maps) or as it would be realistically very difficult to develop housing on certain habitat types. These are coastal habitats (saltmarsh, littoral rock and sediment, supralittoral rock and sediment), inland rock and woodland (coniferous and broadleaf). Therefore, hectares within the 1 km land parcel that are not developable are removed from further calculations.

For each land parcel, we have information on the average house price sold and the proportion of the parcel currently converted to housing. For consistency with the agricultural gross margin data, we transform the house price sold data into a rental value of the land for the developers. To derive this land rental value we first determine the remaining hectares of land within a parcel which can be converted to housing development. We take 100 ha as the standard parcel size and adjust for the total amount of land devoted to urban () and suburban () land use, the remaining land available for development is:

 [9]

Dividing the average house price sold on a land parcel by 100 gives us the average house price sold per hectare, . Using this value and [9] allows us to determine expected house price for the yet undeveloped hectares within a parcel, i.e., the *expected* value of land sold for *potential* housing development:

, [10]

where is the fraction of a house price that accounts for the value of land.

As a final step, we derive the expected *net* development value of land for housing, by adjusting the land rental value obtained in [10] for the gross margin from the agricultural production within the land parcel:

 [11]

We assume that a developer will convert all remaining hectares of a land parcel to housing, subsequently losing any income from agriculture and requiring the purchase of biodiversity offsets. For development to take place, the developer must hold biodiversity offsets equal to the predicted abundance the target species which are lost as a result of converting his land parcel to housing. As a result, the maximum willingness to pay for a single offset for each land parcel is equal to the net value of land for housing [11] relative to the amount of offsets required:

 [12]

We now have a range of offset prices based on farmers’ minimum WTA [5] to change the land management practice and the developers’ maximum WTP [12] for a single offset, from which we can generate the supply and demand curves of offset credits. For each price point, farmer will choose to supply offsets if the supply price is at least equal to the unit cost of providing an offset, i.e., its . This is summarized in the following general decision rule guiding a farmer’s decision:

 [13]again with being the predicted increase in the abundance of the target species.

Similarly, for each price point developers will choose whether or not to purchase offsets, and if so, how many. In this respect, note that a developer must develop a whole parcel and must acquire a sufficient quantity of offset credits to replace all lost species within the parcel. More specifically, the demand for offsets by developer is governed by the following general decision rule:

 [14]

where is the trading ratio as defined in Eq. [1] at time , i.e., prior to changing the land management practice. We can now determine the quantity of offsets that are demanded and supplied at each price point across all land parcels, yielding the offset supply and demand curves and allowing the identification of the offset equilibrium price.

The final stage of the modelling process relates to determining economic welfare as the sum of producer (farmer) and consumer (developer) surplus. Following standard economic principles, we calculate the economic surplus to the housing developers who choose to purchase offsets by subtracting the market clearing price for an offset from their maximum willingness to pay for an offset. We then add up across all participating house builders. Likewise, for farmers who supply offsets, we calculate their producer surplus as the market clearing price for a single offset minus their unit cost to supply a single offset. We then aggregate across all farmers who participate. Given an equilibrium price, , and equilibrium quantity of hectares, total welfare at this market equilibrium then formally reads

 [15]

where the first and second part on the right-hand side is consumer and producer surplus, respectively.

## Model Assumptions

Our empirical framework makes a series of assumptions. Firstly, our model compares two discrete time periods . Period refers to the situation prior to the implementation of the offset market and is based on current land use. Period reflects the state where all possible offset trades have taken place between buyers and sellers at the market equilibrium offset credit price. Beyond this, no further offsetting or development takes place, which ensures no net loss of the ecological target.

Consequently, neither the ecological nor economic models take into account temporal dynamics. From an ecological perspective, this implies that there are no time lags between when we lose an ecologically valuable land parcel to development and where we gain a new ecologically valuable land parcel due to offsetting. In practice, we would expect the ecological losses to take place relatively quickly once the development of a site begins, but the increases in species abundance at the newly created offset site to happen gradually over several years.

A further assumption is that we assume that the current land management practice for farmers (i.e., current crop rotation or livestock) is privately optimal and that switching to a new land management practice will result in a net loss in agricultural profits. We also assume that once land has been secured for biodiversity offsetting it cannot be changed into different land uses and it is protected in perpetuity. We address the implications of these assumptions in the discussion.

A full account of all data sources used, and how these were processed for use in the model, can be found in the Supplementary Information. The authors are very happy to supply the code to run the model on request.

# Results

## Outcomes of Offset Trading

The first design parameter we are interested in is how the choice of the policy target (the unit of exchange), effects the market-clearing price, economic surplus and the ecological landscape. The market-clearing equilibrium price of a credit varies between the three different policy targets (Figure 3 and Table 1). The most expensive offset is for a single oystercatcher which has a market equilibrium price of £21,860 per bird. A single lapwing credit costs £12,600 per bird and a single curlew credit costs £9,971 per bird. At the market equilibrium, there was no net loss in abundance of the policy target for the three different species.

We find that the greatest amount of housing development took place under the oystercatcher policy target with 694 land parcels developed. Of these, 661 land parcels required the developer to purchase offset credits. In contrast, under the lapwing target, 161 land parcels were developed, and 81 required lapwing offset credits to be purchased. Under the curlew target, the smallest amount of development took place, with 108 land parcels developed, of which 61parcels required curlew offset credits to be purchased.

From the supply perspective, there was the highest number of suppliers for the oystercatcher target with 27 agricultural land parcels changing their land management practice to become offset suppliers. Lapwing offsets were supplied by 24 land parcels and curlew offsets were supplied by 6 land parcels.

**[Insert Figure 3 here]**

**[Insert Table 1 here]**

These results are also considered in the spatial context (Figure 4). Figure 4 highlights the land parcels which choose to supply offsets and the land parcels where development that requires offsets takes place. For the curlew policy target, land parcels supplying offsets were located at the coastal margin. In contrast, new development takes places further inland away from the coastal margin. For the lapwing policy target, the offset supply sites for lapwings are geographically close to where the development takes place, with many supply sites within 5 km of where lapwings are due to development. In contrast, curlew supply sites are found at least 15 km from where curlews are lost as a result of new development. Oystercatcher supply sites hold the greatest concentration of offsets, with some of the offset supply sites providing more than 10 offsets per land parcel (which can be thought of as an increase of more than 10 oystercatchers due to the change in land management practice). The oystercatcher supply sites are geographically distant from where oystercatchers are lost as a result of new development, with the greatest distance between offset supply site and development impact over 100 km.

**[Insert Figure 4 here]**

Across all policy targets, there are gains from trade realised as a result of trading biodiversity offsets compared to a no-offset-trading scenario. The greatest gains from trade are realised under the oystercatcher policy target. Across all policy targets, consumer surplus accruing to developers is consistently greater than producer surplus earnt by offset suppliers, suggesting that offset trading offers more benefits to the housing developers than agricultural landowners. Despite these potential gains from trade under the offset market scenario, a small proportion of potential development land was utilised for new developments. The highest number of new developments took place under the oystercatcher policy target. 15% of potentially developable land was developed on compared to 2% of the land area under the lapwing policy target and less than 1% of land area under the curlew policy target.

We can compare the impacts of trading on the non-target species. By this we mean that if the policy target is no net loss of oystercatchers, do we see a positive or negative change in abundance of lapwing and / or curlew (the non-target species in this scenario) as a result of offset trading? The results show, that whilst the biodiversity offset market secures no net loss of the target species under each of the *single* species targets, there is a net loss in abundance of the other two (non-target) species The smallest losses take place under the curlew metric, with a net gain of 21 oystercatchers and a net loss of 14 lapwings. The greatest loss in abundance is under the oystercatcher metric, with a net loss of 3800 lapwings and curlews. This result highlights the differences in habitat preferences of the three species and showcases the complexity and difficulty in defining an offset credit to maintain no net loss of *multiple* species.

## Market Size Effects

The second design parameter we are interested in is how changes to the size of the market, in terms of the geographical area where buyers and sellers can trade, effects the market equilibrium and subsequent economic surplus. We test this by dividing the original case study area into the three smaller service markets, referred to here as Planning areas 1, 2 and 3, which broadly align to the Local Authority planning areas found within the case study region. For this analysis, we focus solely on the oystercatcher policy target. The model using the oystercatcher policy target was thus re-run for each of the three smaller service areas, and compared to results from assuming the whole case study area operates as a single market.

The first result of this analysis showed that market equilibrium was reached in all three planning areas and no net loss of oystercatchers was achieved. Planning area 1 had the lowest cost offset at £771 per oystercatcher, distinctly lower than the clearing price with no trading restrictions (£21,978 per bird at market equilibrium). Within Planning area 1, two land parcels provided a high number of “cheap” oystercatcher offset credits (35 in total) which are sold to 74 developers. Also, a high proportion of development takes place on land parcels which do not require offset credits (33 parcels). The economic surplus for the offset suppliers was approximately 10% of the economic surplus of the developers (£300,000). Comparing development in the full market with the development in Planning area 1, we can see more development takes place in Planning area 1, compared with the same area in the full market (Figure 5).

**[Insert Figure 5 here]**

Planning area 2 had the highest market-clearing price for the oystercatcher offsets at £39,632 per bird. 32 land parcels became oystercatcher supply sites, which in total supplied 41 oystercatcher credits (on average 1.3 oystercatchers supplied per parcel). This is significantly smaller than the number of offsets supplied per land parcel compared to Planning Areas 1 and 3 (on average 17.5 per parcel and 8.1 per parcel respectively). Compared to the model with no restrictions on trading, there is significantly less development taking place in Planning area 2, as a result of the increase in the price of an oystercatcher credit.

Planning area 3 had the greatest number of land parcels requiring an offset (302), with a market-clearing price of £26,699 per oystercatcher credit. The resulting distribution of housing developments in Planning area 3 is broadly the same as results of the single market model. This similarity is a result of Planning area 3 containing the majority of the land parcels which supply offsets in the single market scenario.

We can compare the economic surplus generated for the three planning areas with the surplus generated for the market with no geographic trading restrictions. Restricting trading within the planning regions offers a small increase in the total surplus for offset suppliers of approximately £8000. This is due to the increase in the additional land parcels supplying offsets in Planning area 3 and the increase in market clearing price for Planning areas 2 and 3 compared to the no restriction model. In contrast, the surplus for housing developers reduces by £5,000 (Figure 6).

**[Insert Figure 6 here]**

From an ecological viewpoint, restricting the spatial area for trading is beneficial to the non-target species (curlews and lapwings). Under the trading with no geographical restrictions, there was a net loss of 3800 curlews and lapwing compared to a net loss of 2200 curlew and lapwing under the restricted trading. This was due to the geographical redistribution of offset trades. Restricting the size of the market thus seems to have ecological benefits.

# Discussion

Tradeable offset credits offer a promising approach to tackling the growing global conflicts between land development and biodiversity conservation. Whilst there has been an increase in policy interest in the concept, and a large ecological literature studying the problem, there have been relatively few contributions from economists to thinking about how best to design such offset markets. Our paper is designed to contribute to this gap in the literature. One key contribution of our paper is a modelling framework from which we can derive spatially explicit supply and demand curves for biodiversity offsets. These allow us to test how changes to two offset market design parameters: what is traded (the ecological target) and where it is traded (the size of the market), affect the functioning of the market. To do this, we develop and integrate ecological and economic models which encapsulate the spatial heterogeneity in the biodiversity potential of individual sites as well as spatial heterogeneity in supply and demand prices.

Results show that the choice of the ecological policy target has a significant effect on the number of trades taking place, the amount of new housing development, the unintended ecological consequences on non-target species, and economic welfare. Each of the three species considered as no net loss targets favour different land parcels, as predicted by the ecological model. From a supply perspective, this means that each ecological target has a varying opportunity cost in terms of agricultural land foregone. From a demand perspective, land parcels with development potential have varying permit requirements depending on which species is chosen as the no net loss target, whilst house builders’ maximum Willingness to Pay for permits varies spatially with housing values.

Using the spatially derived supply and demand curves reveals that the market does not always react according what one might expect a priori. The curlew policy target had the lowest equilibrium offset price of £9,972 per bird yet resulted in the *fewest* number of trades taking place (10 trades and 108 land parcels developed). In contrast, the no net loss target which resulted in the costliest equilibrium offset price (£21,979 per oystercatcher) had the highest number of trades (260) with 694 land parcels developed. Based on our comparative static analysis, we might have presumed that the species which offered the cheapest offsets would have led to the most development taking place as more developers would have been willing to purchase a cheaper offset. This follows from an assumption that cheaper credit prices would reduce opportunity costs for developers. However, based on our *integrated* modelling this result is more nuanced: the greatest abundance of curlews and lapwings are found in the areas with the highest development values, and subsequently, a greater number of credits is required by the developer per land parcel developed. As such, developers are willing to pay less per offset credit. Moreover, the most favourable areas for lapwing and curlew supply are in areas where the value of agricultural output is higher (and so the opportunity cost of creating credits is higher), increasing the individual farmers' unit cost of offset provision. There are very few farmers who offer offsets to the market below the maximum willingness to pay of the housing developer. The market is “constrained,” imposing a downward pressure on the offset price to a level below what many suppliers would be willing to accept.

In contrast, oystercatchers are recorded at lower abundances in the areas of high housing value, meaning developers are required to spend less on offsetting per parcel, increasing their maximum willingness to pay for a single offset and thus encouraging more farmers to supply oystercatcher offsets. In addition, under the full market size scenario, all oystercatcher offsets are created at the coastal margin where the value of agricultural land is low but the potential for the number of new offsets to be generated is high. This influences the farmers’ willingness to accept payment for switching from agriculture to offset generation.

Our results also add further evidence to the argument deciding what unit of biodiversity to trade is inherently complex. For each of our models, we focus on ensuring no net loss of the target species and this condition is always met. However, we also analyse the impact on the other two species. Across all three policy targets, we see a net loss in the bird species not protected by the metric adopted. This follows from the fact that the three species have different habitat requirements. Developing a market that would ensure no net loss of these three species *simultaneously* would, therefore, be more complex and likely depress the level of new housing development, although this would be ecologically more beneficial. Simple single species metrics lend themselves to simple trading rules, and thus lower transactions costs, but can result in negative impacts on other measures of biodiversity.

In terms of economic welfare, results show there is a systematically large difference between consumer and producer surplus. Consumer surplus is consistently greater across the three policy targets, indicating that the market benefits housing developers more than farmers. This has an interesting relevance given the current state of biodiversity offset discussions within the UK policy context. The current plans are backed heavily by housing developers, with many even keen to adopt a “net gain agenda”. This approach to development that aims to leave the natural environment in a measurably better state than beforehand (HM Government, 2018). However, there is a significant lack of farmers/landowners willing to take part in the planning process or engage in the biodiversity offset pilot schemes (Sullivan and Hannis, 2015). At present, the current incentives offered by biodiversity offsetting are not strong enough for these entities to begin to engage with the policy process.

Our analysis also considered how changing the geographical market region affects economic and ecological outcomes. In general, efficient markets are those featuring a high number of participants in order to stimulate trading and enhance market liquidity. Such markets would likely feature less volatile prices than thin markets, reducing price uncertainty in a manner conducive to more investment in conservation (Wissel and Wätzold, 2010). Our theoretical model predicts that an increase in the number of sellers would increase the number of hectares that generate offsets, resulting in downward pressure on the equilibrium credit price. In contrast, the equilibrium price would increase if demand would be enhanced through an increase in the number of buyers.

In our empirical model, we can only adjust the number of potential sellers by changing the market boundaries. However, by doing this we change the spatial heterogeneity embedded in both the supply and demand curves. This leads to a counter-intuitive empirical result when we repeat our analysis over three restricted trading regions (referred to as Planning areas). The smallest region for trading (Planning area 1) had the lowest equilibrium price offset credit. This region only contained two offset supply sites, which, however, offered a high number of credits due to the high ecological productivity of the two land parcels (that is, changing the land management to grassland produced a large predicted rise in target species abundance). Since credits are supplied relatively cheaply due to the low opportunity costs of the land, the total amount of development increases within this area (for new development not requiring offsets) due to spatial heterogeneity in land value for development. That is, land value in the region is low, thus depressing the developers’ maximum willingness to pay for an offset. As such, when trading could take place across the whole case study area un-restricted, and the market equilibrium offset price was higher, the development would not take place in the same locations since developers would instead choose more profitable land parcels.

In contrast, for Planning areas 2 and 3, the credit prices are greater than the unrestricted trading scenario market equilibrium price of (£26,000 and £29,000 per offset respectively). In these two regions where the credit price increases, demand for new development falls, consistent with the theoretical prediction. Spatially this is due to these areas no longer having access to the cheapest offset supply sites. This is particularly evident in Planning area 3 where new offset suppliers enter the market as a result of an increase in the equilibrium offset supply price. Ecologically it appears that configuring smaller trading regions is more beneficial, as the losses in the non-target species are smaller than the full market scenario. This is a result of the decline in new development due to the increase in credit price in two of the restricted planning regions.

Another ecological concern of trading taking place across the full case study area is the spatial differences between the location of newly created offset sites and areas of land lost to development. For example, under the oystercatcher policy target, development takes place in upland regions where oystercatchers breed in the spring and summer months, but offset sites are created along the coastal margin, a favourable overwintering habitat. A benefit of more spatially-restricted markets is that this forces the creation of more expensive offset sites inland (creating potential breeding habitat). If unrestricted trading was preferred across the whole case study area, a trading rule where only spring /summer habitat could be traded for newly created spring /summer habitat could be specified. This likely reduces the total amount of new development due to higher credit prices.

We would also like to note some limitations in the model and its application. Firstly, our model does not consider intertemporal dynamics regarding the ecological and economic aspects of the market. In our modelling approach, we chose to model two states: current land use where no offsetting took place and a second state where all offsetting takes place up to where the market reaches an equilibrium. From an ecological viewpoint, this does not consider dynamics and time lags in the generation of the offset credits related to the target species, which is a key concern of ecologists regarding offsetting (Gibbons and Lindenmayer, 2007; Maron et al, 2012). As recommended in Needham et al, (2019), if offset markets are to be developed then we would recommend the use of “banked credits” where trading can only take place once the credits have been certified as providing an increase in the target species, which seeks to overcome some of the problems associated with uncertainty in restoration.

From an economics perspective, we do not consider dynamics associated with land value changes associated with new housing development. The rental value of land may decrease once a threshold of development on neighbouring land parcels has been reached, thus reducing potential profits for developers and reducing the derived demand for credits. We also assume that once land has been secured for through an offset it cannot be changed into a different land use, i.e., it is protected in perpetuity. Evidence from the UK biodiversity offset pilots shows that landowners are strongly opposed to this policy, preferring a 10 year time horizon for offset contracts (Hannis and Sullivan, 2015). This would likely lead to a net loss of the ecological target over time and undermines the potential of offset markets to deliver biodiversity protection. Stronger incentives would need to be offered to landowners to encourage them to set aside their land permanently.

The supply prices of offsets were determined solely by agricultural gross margin data. We acknowledge that this approach does not consider the social implications of offset developments, for example, loss of locally important offset areas and access to green spaces. This is an aspect of offsetting that is being discussed in greater detail in the literature, particularly with respect to offsetting in developing countries (Griffiths et al, 2019). We also recognise that there were many alternative land management strategies which could have been pursued within the modelling framework to generate biodiversity offset credits. The restoration of heather and heather grassland and saltmarsh which would have been expected to deliver greater increases in the target species than changing agricultural land use to improved grassland in order to create credits. We note that the restoration of certain habitats would incur restoration costs for the farmer and these would need to be captured in the credit price, thus changing the market equilibrium and subsequently the resulting economic and ecological impacts. This is something we would recommend pursuing in further work.

We also assumed that trading was mandatory across the whole case study region, with any development impact on the ecological metric requiring an offset. This is in contrast to current offset schemes globally where the purchasing of offsets is often undertaken voluntarily by developers, who can also choose to undertake offsetting in the form of compensatory mitigation directly rather than purchasing offsets through a market. This has been shown to reduce the demand for third-party generated offsets. This not only suppresses the development of an offset trading scheme but in many cases also leads to lower quality restoration actions than those which would have been undertaken through a regulated offset supplier such as an offset bank (Bonnie 1999; Bekessy et al, 2010).

The analytical framework here could be used to further study various design parameters within biodiversity offset markets. For example, different ecological metrics could be used, such as no net loss of habitats or an overall measure of species richness; temporal relationships could be explored through the use of a dynamic model that allows for feedbacks in the system to account for impacts on surrounding land parcels when adjacent parcels are either developed or certain land management practices are changed.

# Conclusions

This study provides a novel analysis of biodiversity offset markets, integrating ecological species abundance modelling with economic modelling of market outcomes. Our paper has provided evidence on the importance of the choice of the ecological metric for trading, as determined by the policy target, and of the size of the market for trading, in terms of the development of a biodiversity offset market. Empirical modelling of such offset markets has shown that altering these two design parameters does not necessarily affect the market in expected ways. The choice of metric affects both the scale and distribution of new development, as well as the ecological gains and losses, which differed significantly between our three no net loss target species. This adds further evidence to the argument that defining a credit for biodiversity offset trading is inherently complex. In addition, the cheapest offset credit leads to the lowest amount of new development, and conversely, the most expensive credits were associated with the greatest amount of new development.

Regarding market size, we show that a larger geographic region for trading benefits developers. This greater scale allows each developer to access the cheapest offset supply sites. Restricting trading to only being allowed to take place within a smaller planning region constrains developers’ ability to access to the cheapest offsets and exerts downward pressure on new development. However, restricting trading to smaller areas has ecological benefits for both target and non-target species, and a policy designer would need to balance these against the economic impacts of restricting trading in deciding whether, and how, to segment an offset market.

Whilst the results that emerge from our modelling on equilibrium prices, surplus changes, ecological impacts and housing development are of course specific to the case study area considered, some generalizable messages are also available. The first is that the outcomes of a market in biodiversity offsets seem to depend crucially on the spatial correlation of ecological potential, the opportunity costs of offset creation, and housing land values. In our study, what seem as “counter-intuitive” results largely emerge from a specific spatial correlation pattern of these variables. Second, changing market design parameters such as market size, ecological metric and trading rules are not likely to have equivalently signed implications for ecological and economic outcome measures. Changes which improve ecological outcomes may well worsen economic outcomes. Thirdly, given the multi-faceted nature of biodiversity, choice of a specific offset metric (in this case, which bird species is chosen) may have unintended negative effects on other important aspects of biodiversity, such as ecosystem functioning, resilience and species impacts.

In conclusion, our work has provided evidence to help understand how the choice of the ecological target and the size of the market affect the development and functioning of a biodiversity offset market. Implementing such markets could assist in reducing development-conservation conflicts. However, offsetting alone is not sufficient to guarantee no declines in wider measures of biodiversity, and existing national and international biodiversity protection must remain in place (such as Special Protection Areas, for example). Instead, offsetting has the potential to reduce the loss of biodiversity for ecologically valuable habitats that are not within the internationally protected regions as part of a package of measures, with the aim of moving towards a net gain in nation-wide biodiversity.

# Acknowledgements

We thank The Leverhulme Trust for funding this work under project RPG-2017-148 and members of our advisory group for numerous helpful comments on the research.

# References

Atkinson, S. E., & Tietenberg, T. H. 1987. “Economic implications of emissions trading rules for local and regional pollutants.” *Canadian Journal of Economics*, 370-386.

BBOP. 2009. Business and Biodiversity Offsets Programme (BBOP). 2009. Biodiversity Offset Design Handbook. BBOP, Washington, D.C.

Barker, Nicole K. S., Stuart M. Slattery, Marcel Darveau, and Steve G. Cumming. 2014. “Modeling Distribution and Abundance of Multiple Species: Different Pooling Strategies Produce Similar Results.” *Ecosphere* 5 (12): art158. https://doi.org/10.1890/ES14-00256.1.

Balmer, D. E., Gillings, S., Caffrey, B. J., Swann, R. L., Downie, I. S., & Fuller, R. J. 2013. Bird Atlas 2007-11: the breeding and wintering birds of Britain and Ireland. Thetford: BTO.

Beattie, Alastair. 2019. *The Farm Management Handbook 2019/20*. SRUC. www.fas.scot.

Bekessy, Sarah A., Brendan A. Wintle, David B. Lindenmayer, Michael A. Mccarthy, Mark Colyvan, Mark A. Burgman, and Hugh P. Possingham. 2010. “The Biodiversity Bank Cannot Be a Lending Bank.” *Conservation Letters* 3 (3): 151–58. https://doi.org/10.1111/j.1755-263X.2010.00110.x.

Bendor, Todd. 2009. “A Dynamic Analysis of the Wetland Mitigation Process and Its Effects on No Net Loss Policy.” *Landscape and Urban Planning* 89 (1–2): 17–27. https://doi.org/10.1016/j.landurbplan.2008.09.003.

BenDor, Todd K., Tianshu Guo, and Andrew J. Yates. 2014. “Optimal Advanced Credit Releases in Ecosystem Service Markets.” *Environmental Management* 53 (3): 496–509. https://doi.org/10.1007/s00267-013-0219-1.

BenDor, Todd., & Nicholas Brozović. (2007). Determinants of spatial and temporal patterns in compensatory wetland mitigation. *Environmental Management* 40 349– 364. <https://doi.org/10.1007/s00267-006-0310-y>

BirdLife International (2020) Species factsheet: *Numenius arquata*. Downloaded from [http://www.birdlife.org](http://www.birdlife.org/) on 24/07/2020.

Bonds, Matthew H., And Jeffrey J. Pompe. 2003. “Calculating Wetland Mitigation Banking Credits: Adjusting for Wetland Function and Location.” *Natural Resources Journal*. Regents of the University of New Mexico on behalf of its School of Law. https://doi.org/10.2307/24888894.

Bonnie, Robert. 1999. “Endangered Species Mitigation Banking: Promoting Recovery through Habitat Conservation Planning under the Endangered Species Act.” *Science of the Total Environment* 240 (1–3): 11–19. https://doi.org/10.1016/S0048-9697(99)00315-0.

Brotons, Lluís, Wilfried Thuiller, Miguel B Araújo, Alexandre H Hirzel Brotons, Araú Jo, M B Hirzel, and / W Thuiller. 2004. “Presence-Absence versus Presence-Only Modelling Methods for Predicting Bird Habitat Suitability.” *ECOGRAPHY* 27: 4.

Bull, J.W., Suttle, K.B., Gordon, A., Singh, N.J. & Milner-Gulland, E.J. 2013. "Biodiversity offsets in theory and practice". *Oryx,* 47, 369-380

Bull, Joseph W., Samuel P. Lloyd, and Niels Strange. 2017. "Implementation gap between the theory and practice of biodiversity offset multipliers." *Conservation Letters* 10 (6): 656-669.

Burnham, Kenneth P., and David R. Anderson. 1998. "Practical use of the information-theoretic approach." In *Model selection and inference*, pp. 75-117. Springer, New York, NY,

Coralie, Calvet, Ollivier Guillaume, and Napoleone Claude. 2015. “Tracking the Origins and Development of Biodiversity Offsetting in Academic Research and Its Implications for Conservation: A Review.” *Biological Conservation*. Elsevier Ltd. https://doi.org/10.1016/j.biocon.2015.08.036.

Stockton-on-Tees Borough Council, 2016. “Economic Growth Plan.” http://egenda.stockton.gov.uk/aksstockton/images/att30085.pdf.

Doyle, Martin W., and Andrew J. Yates. 2010. “Stream Ecosystem Service Markets under No-Net-Loss Regulation.” *Ecological Economics* 69 (4): 820–27. https://doi.org/10.1016/j.ecolecon.2009.10.006.

Doyle, M. W., Patterson, L. A., Chen, Y., Schnier, K. E., and Yates, A. J. 2014. Optimizing the scale of markets for water quality trading, *Water Resour. Res*., 50, 7231– 7244,

Drechsler, Martin, and Florian Hartig. 2011. “Conserving Biodiversity with Tradable Permits under Changing Conservation Costs and Habitat Restoration Time Lags.” *Ecological Economics* 70 (3): 533–41. https://doi.org/10.1016/j.ecolecon.2010.10.004.

Fernandez, Linda, and Larry Karp. 1998. “Restoring Wetlands through Wetlands Mitigation Banks.” *Environmental and Resource Economics* 12 (3): 323–44. https://doi.org/10.1023/A:1008225021746.

Ferrari, Edward, and Karl Dalgleish. 2019. “Tees Valley Local Housing Markets.” https://www4.shu.ac.uk/research/cresr/sites/shu.ac.uk/files/tees-valley-local-housing-markets.pdf

Fisher‐Vanden, K., & Shelia Olmstead, S. 2013. Moving pollution trading from air to water: Potential, problems, and prognosis. *Journal of Economic Perspectives* 27: 147– 172. <https://doi.org/10.1257/jep.27.1.147>

Franks, Samantha E., David J.T. Douglas, Simon Gillings, and James W. Pearce-Higgins. 2017. “Environmental Correlates of Breeding Abundance and Population Change of Eurasian Curlew Numenius Arquata in Britain.” *Bird Study* 64 (3): 393–409. https://doi.org/10.1080/00063657.2017.1359233.

Harris, S. J., D. Massimino, S. E. Newson, M. A. Eaton, D. E. Balmer, D. G. Noble, A. J. Musgrove, S. Gillings, D. Procter, and J. W. Pearce-Higgins. 2015. "The Breeding bird survey 2014." *BTO research report* 673 (2015).

HM Government 2018. “Land Value Estimates.” 2018. https://www.gov.uk/government/collections/land-value-estimates.

Griffiths, Victoria F., Joseph W. Bull, Julia Baker, and E. J. Milner-Gulland. 2019. “No Net Loss for People and Biodiversity.” *Conservation Biology* 33 (1): 76–87. https://doi.org/10.1111/cobi.13184.

Hannis, Mike, and Sian Sullivan. n.d. *Offsetting Nature? Habitat Banking and Biodiversity Offsets in the English Land Use Planning System*. Accessed February 3, 2020. http://www.greenhousethinktank.org.

Heal, Geoffrey M. 2005. “Arbitrage, Options and Endangered Species.” *SSRN Electronic Journal*, November. https://doi.org/10.2139/ssrn.457330.

Hewson, C.M., Thorup, K., Pearce-Higgins, J.W. and Atkinson, P.W., 2016. "Population decline is linked to migration route in the Common Cuckoo". *Nature Communications,* 7(1), pp.1-8.

JNCC, Joint Nature Conservation Committee). 2020. “Special Protection Areas – Overview | JNCC - Adviser to Government on Nature Conservation.” Special Protection Areas – Overview. 2020. https://jncc.gov.uk/our-work/special-protection-areas-overview/.

Kangas, Johanna, and Markku Ollikainen. 2019. “Economic Insights in Ecological Compensations: Market Analysis With an Empirical Application to the Finnish Economy.” *Ecological Economics* 159 (May): 54–67. https://doi.org/10.1016/j.ecolecon.2019.01.003.

Laitila, Jussi, Atte Moilanen, and Federico M. Pouzols. 2014. “A Method for Calculating Minimum Biodiversity Offset Multipliers Accounting for Time Discounting, Additionality and Permanence.” *Methods in Ecology and Evolution* 5 (11): 1247–54. https://doi.org/10.1111/2041-210X.12287.

Lapeyre, Renaud, Géraldine Froger, and Marie Hrabanski. 2015. “Biodiversity Offsets as Market-Based Instruments for Ecosystem Services? From Discourses to Practices.” https://doi.org/10.1016/j.ecoser.2014.10.010.

Levrel, Harold, Pierre Scemama, and Anne-Charlotte Vaissière. 2017. Should we be wary of mitigation banking? Evidence regarding the risks associated with this wetland offset arrangement in Florida. *Ecological Economics* 135: 136–149. https://doi.org/10.1016/j.ecolecon.2016.12.025

Magurran, Anne E., Stephen R. Baillie, Stephen T. Buckland, Jan McP Dick, David A. Elston, E. Marian Scott, Rognvald I. Smith, Paul J. Somerfield, and Allan D. Watt. 2010. "Long-term datasets in biodiversity research and monitoring: assessing change in ecological communities through time." *Trends in ecology & evolution* 25 (10) : 574-582.

Maron, Martine, Richard J. Hobbs, Atte Moilanen, Jeffrey W. Matthews, Kimberly Christie, Toby A. Gardner, David A. Keith, David B. Lindenmayer, and Clive A. McAlpine. 2012. “Faustian Bargains? Restoration Realities in the Context of Biodiversity Offset Policies.” *Biological Conservation* 155 (October): 141–48. https://doi.org/10.1016/j.biocon.2012.06.003.

McKenney, Bruce A., and Joseph M. Kiesecker. 2010. “Policy Development for Biodiversity Offsets: A Review of Offset Frameworks.” *Environmental Management*. https://doi.org/10.1007/s00267-009-9396-3.

Needham, Katherine, Frans P de Vries, Paul R Armsworth, and Nick Hanley. 2019. “Designing Markets for Biodiversity Offsets: Lessons from Tradable Pollution Permits.” *Journal of Applied Ecology* 56 (6): 1429–35. https://doi.org/10.1111/1365-2664.13372.

Northern Upland Chain Local Nature Partnership. 2020. Curlew conservation. Online. Webpage available at: <https://www.nuclnp.org.uk/>

Pearce-Higgins, J.W. and Crick, H.Q.P., 2019. "One-third of English breeding bird species show evidence of population responses to climatic variables over 50 years". *Bird Study*, 66(2), pp.159-172.

Potts, Joanne M., and Jane Elith. 2006. “Comparing Species Abundance Models.” *Ecological Modelling* 199 (2): 153–63. https://doi.org/10.1016/j.ecolmodel.2006.05.025.

Renwick, A.R., Massimino, D., Newson, S.E., Chamberlain, D.E., Pearce‐Higgins, J.W. and Johnston, A., 2012. "Modelling changes in species’ abundance in response to projected climate change". Diversity and Distributions, 18(2), pp.121-132.

Rowland, C.S., Morton, R.D., Carrasco, L., McShane, G., O'Neil, A.W., Wood, C.M. 2017. Land Cover Map 2015 (vector, GB). NERC Environmental Information Data Centre https://doi.org/10.5285/6c6c9203-7333-4d96-88ab-78925e7a4e73

Simpson, Katherine. 2011. “Implementing an Ecosystem Approach:The Case of the Teesmouth and Cleveland Coast European Marine Site.”

Smith, Ken. 2011. “The State Of The Natural Environment Of The Tees Estuary A Review Of The Bird Chapter.” http://www.inca.uk.com/wp-content/uploads/2012/01/BIRDS-SONET-UPDATE-2011.pdf.

Sullivan, S., and M. Hannis. 2015. “Nets and Frames, Losses and Gains: Value Struggles in Engagements with Biodiversity Offsetting Policy in England.” *Ecosystem Services* 15 (October): 162–73. https://doi.org/10.1016/j.ecoser.2015.01.009.

Tattoni, Clara, Franco Rizzolli, and Paolo Pedrini. 2012. “Can LiDAR Data Improve Bird Habitat Suitability Models?” *Ecological Modelling* 245 (October): 103–10. https://doi.org/10.1016/j.ecolmodel.2012.03.020.

Ten, Kerry, Kate Jo Treweek, and Jon Ekstrom. 2010. “The Use Of Market-Based Instruments For Biodiversity Protection-The Case Of Habitat Banking European Commission DG Environment.” www.eftec.co.uk.

Venables, W N, and B D Ripley Springer. n.d. “Modern Applied Statistics with S Fourth Edition.” Accessed February 3, 2020. http://www.insightful.com.

Vries, Frans P. de, and Nick Hanley. 2016. “Incentive-Based Policy Design for Pollution Control and Biodiversity Conservation: A Review.” *Environmental and Resource Economics*. Springer Netherlands. https://doi.org/10.1007/s10640-015-9996-8.

Wissel, Silvia, and Frank Wätzold. 2010. “A Conceptual Analysis of the Application of Tradable Permits to Biodiversity Conservation” *Conservation Biology* 24 (2): 404–11. https://doi.org/10.1111/j.1523-1739.2009.01444.x.

Womble, Philip, and Martin Doyle. 2012. "The geography of trading ecosystem services: A case study of wetland and stream compensatory mitigation markets." *Harv. Envtl. L. Rev.* 36: 229.

**Table 1: A comparison of the market equilibrium prices for the offset policy targets**

|  |  |  |  |
| --- | --- | --- | --- |
|   | Curlew policy target | Lapwing policy target | Oystercatcher policy target |
| Market equilibrium price for a single offset credit | £9,972 | £12,600 | £21,979 |
| Number of developers requiring offsets  | 45 | 87 | 661 |
| Economic surplus for developers | £421,885 | £404,865 | £1,240,000 |
| Number of farmers supplying offsets | 6 | 24 | 28 |
| Number of offsets supplied (birds) | 10 | 103 | 260 |
| Economic surplus for farmers | £27,661 | £52,458 | £480,240 |

**Figure 1: Overview of the ecological-economic modelling approach**

**Figure 2: UK case study region. Contains OS data 1:250 000 Scale Colour Raster © Crown Copyright 2019.**

**Figure 3: Spatially derived offset supply and demand curves for the three target species. Dashed line indicates market equilibrium price, p.\***

**Figure 4: Comparison of land parcels which deliver biodiversity offsets and which land parcels require offset credits for the three target species.**

**Figure 5: A comparison of offset supply and offset demand in the three planning areas for oystercatcher as the policy target.**

**Figure 6: Comparison of the economic surplus for farmers and developers at the market equilibrium offset price, p\***

1. This could mean that some developers choose to buy no credits and not develop any housing. [↑](#endnote-ref-1)
2. If there is more than one ecological indicator, then there are multiple trading ratios between sites, one for each target. [↑](#endnote-ref-2)
3. In a similar vein, the opposite effect occurs when the trading ratio as defined in Eq. (1) decreases through either a decrease (increase) in the ecological value of site $i$ ($j$). In both these situations the developer’s demand for offset credits increases, pushing up the equilibrium offset price. [↑](#endnote-ref-3)
4. The *Walrasian auctioneer* is the presumed auctioneer that matches supply and demand in a market of perfect competition. The auctioneer provides for the features of perfect competition: perfect information and no transaction costs. The process is called *tâtonnement*, relating to finding the market clearing price for all commodities (Joyce, 1984). [↑](#endnote-ref-4)
5. Note that whilst the farmer will lose the gross margin for the original crop (the opportunity cost), still a profit on the replacement crop will be made after switching. [↑](#endnote-ref-5)