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CONTRIBUTED PAPER

Ecological and economic implications of alternative metrics in biodiversity offset markets

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Article Impact Statement: Policies should offer the highest incentives for conserving and enhancing the most ecologically beneficial sites in a landscape.

Abstract

Policy tools are needed that allow reconciliation of human development pressures with conservation priorities. Biodiversity offsetting can be used to compensate for ecological losses caused by development activities. Landowners can choose to undertake conservation actions, including habitat restoration, to generate biodiversity offsets. Consideration of the incentives facing landowners as potential biodiversity offset providers and developers as potential buyers of credits is critical when considering the ecological and economic landscape-scale outcomes of alternative offset metrics. There is an expectation that landowners will always seek to conserve the least profitable land parcels, and, in turn, this determines the spatial location of biodiversity offset credits. We developed an ecological-economic model to compare the ecological and economic outcomes of offsetting for a habitat-based metric and a species-based metric. We were interested in whether these metrics would adequately capture the indirect benefits of offsetting on species not considered under a no-net-loss policy. We simulated a biodiversity offset market for a case study landscape, linking species distribution modeling and an economic model of landowner choice based on economic returns of the alternative land management options (restore, develop, or maintain existing land use). Neither the habitat nor species metric adequately captured the indirect benefits of offsetting on related habitats or species. The underlying species distributions, layered with the agricultural and development rental values of parcels, resulted in very different landscape outcomes depending on the metric chosen. If policy makers are aiming for the metric to act as an indicator to mitigate impacts on a range of closely related habitats and species, then a simple no-net-loss target is not adequate. Furthermore, to achieve the most ecologically beneficial design of offsets policy, an understanding of the economic decision-making processes of the landowners is needed.

KEYWORDS
biodiversity loss, biodiversity metrics, biodiversity offsets, no net loss, simulation model

Resumen

Se necesitan herramientas políticas que permitan la reconciliación entre las presiones del desarrollo humano y las prioridades de conservación. La compensación de biodiversidad puede usarse para reponer las pérdidas ecológicas causadas por las actividades de desarrollo. Los terratenientes pueden elegir realizar acciones de conservación, incluyendo la restauración del hábitat, para generar dichas compensaciones. Es importante considerar los incentivos para los terratenientes como proveedores potenciales de compensaciones de biodiversidad y para los desarrolladores como compradores potenciales de créditos cuando se contemplan los resultados ecológicos y económicos a escala de paisaje de estas medidas alternativas de compensación. Existe la expectativa de que los terratenientes siempre buscarán conservar los lotes menos rentables y, por lo tanto, esto determina la ubicación

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espacial de los créditos por compensación de biodiversidad. Desarrollamos un modelo para comparar los resultados ecológicos y económicos de la compensación en una medida basada en el hábitat y una basada en la especie. Nos interesaba saber si estas medidas indicarían adecuadamente los beneficios indirectos de la compensación para las especies no consideradas bajo una política de pérdida neta cero. Simulamos un mercado voluntario de biodiversidad para un estudio de casode un paisaje, el cual vinculó el modelado de la distribución de especies con el modelo económico de las elecciones de los terratenientes basadas en las ganancias económicas de las opciones alternativas de manejo de suelo (restaurar, desarrollar o mantener el uso de suelo existente). Ninguna de las dos medidas indicó adecuadamente los beneficios indirectos de la compensación para las especies o hábitats relacionados. La distribución subyacente de especies, en conjunto con los valores de renta agrícolas y de desarrollo de los lotes, derivó en resultados muy diferentes de paisaje según la medida seleccionada. Cuando los formuladores de políticas buscan que la medida actúe como un indicador para mitigar impactos en una gama de especies y hábitats relacionados cercanamente, no es adecuado un objetivo simple de pérdida neta cero. Además, para lograr el diseño con el mayor beneficio ecológico, se requiere comprender los procesos de decisión de los terratenientes.

PALABRAS CLAVE:
compensaciones de biodiversidad, medidas de biodiversidad, modelo de simulación, pérdida de biodiversidad, pérdida neta cero

生物多样性补偿市场中替代指标的生态和经济影响

【摘要】人类发展压力与保护优先事项之间的平衡需要政策工具来调解。生物多样性补偿可以用于弥补发展活动造成的生态损失。土地所有者可以选择采取包括生境恢复在内的保护行动, 来补偿生物多样性的丧失。在评估替代性补偿指标在景观尺度上的生态及经济效益时, 考虑到土地所有者作为潜在的生物多样性补偿提供者和开发者作为潜在的信用购买者所面临的激励措施是至关重要的。一般认为, 土地所有者总是会寻求保护利润最低的地块, 反过来, 这也决定了生物多样性补偿信用的空间位置。本研究开发了一个生态经济模型, 以比较基于生境的指标和基于物种的指标得到的生物多样性补偿的生态和经济结果。我们想探究这些指标是否能充分捕捉到无净损失政策未考虑到的物种在生物多样性补偿中获得的间接利益。为此, 我们模拟了一个案例研究景观中的生物多样性补偿市场, 将物种分布模型与土地所有者基于替代土地管理方案 (恢复、开发或维持现有土地使用) 经济回报做出决策的经济模型相结合, 结果表明, 生境指标和物种指标都不能充分捕捉到生物多样性补偿对相关生境或物种的间接效益。潜在的物种分布加上地块的农业和开发租赁价值, 根据所选指标会导致全然不同的景观结果。如果政策制定者的目标是基于指标评估来减缓对一系列密切相关的生境和物种的影响, 那么简单的无净损失目标是不够的。此外, 为了实现最有利于生态的补偿政策设计, 还需要了解土地所有者的经济决策过程。**【翻译: 胡怡思; 审校: 聂永刚】**

关键词: 生物多样性补偿, 生物多样性指标, 无净损失, 生物多样性丧失, 模拟模型

INTRODUCTION

Goal 15 of the UN Sustainable Development Goals is to halt and reverse land degradation and the associated loss of biodiversity (United Nations, 2015). However, the human population is predicted to reach 8.6 billion by 2030, an increase of 1 billion from 2020 (United Nations, 2017). Consequently, ceasing human development impacts (including new housing and infrastructure) is not an option (United Nations, 2019). Instead, tools are needed that allow the reconciliation of development pressures with biodiversity conservation. Biodiversity offsets are one such policy option that is being increasingly applied to respond to these pressures (Moilanen & Kotiaho, 2021).

Biodiversity offsets provide "measurable conservation outcomes resulting from actions designed to compensate for significant residual adverse biodiversity impacts" (BBOP, 2009). Offsetting is considered the final step in the mitigation hierarchy once all other steps (avoid, minimize, and restore) have been undertaken (Arlidge et al., 2018). The majority of offset policies target no net loss of biodiversity, where losses due to development are matched through gains in biodiversity elsewhere (Zu Ermgassen et al., 2019). More recently, the focus has been shifting toward net positive impact and biodiversity net gain, which aim to improve the state of the environment relative to the predevelopment state (Bull & Brownlie, 2017; Moilanen & Kotiaho, 2021; McVittie & Faccioli, 2020).

We focused on markets for biodiversity offsets. These markets are created when multiple buyers and sellers of offsets interact with others through a trading process, typically moderated by an offset bank or regulator (Needham et al., 2019). Landowners can choose to manage land for conservation, generating offset credits that can then be sold to a developer who is required to mitigate development impacts, for example, from house building, on some measure of biodiversity. By establishing an appropriate rate of exchange between sellers and buyers, markets can, in theory, achieve no net loss of biodiversity (or a net gain) within some defined area at least cost.

One of the most contentious issues in the design of offsetting schemes is the choice of the offset metric: how gains and losses in biodiversity are assessed and compared. This metric forms the trading unit within an offset market. Across the disciplines of economics and ecology, the choice of metric is seen as critical in determining the success of offsetting as a policy instrument (Bull et al., 2013; Heal, 2005). From an economic perspective, markets require goods to be grouped into simple, measurable, standardized units to foster exchangeability and market efficiency (Salzman & Ruhl, 2001). However, it is difficult to translate biodiversity into a simple metric as part of a market exchange mechanism (Bull et al., 2013). Many of the most popular offset metrics use a combination of habitat area, vegetation, and site condition scores (Bull et al., 2014; Zu Ermgassen, 2019). There is an expectation from the policy community that these metrics will adequately capture many of the indirect benefits of offsetting, such as increasing the numbers of other, nontarget plant and animal species (Cristescu et al., 2013; Marshall et al., 2020a). However, the evidence thus far has demonstrated that these approaches rarely achieve no net loss of biodiversity (Maron et al., 2012; Bull et al., 2014; Zu Ermgassen et al., 2019).

Alternative offset metrics include more detailed species data and compare their ecological outcomes with habitat-based metrics (Marshall et al., 2020; Maseyk et al., 2016; McVittie & Faccioli, 2020b). However, there has been little quantitative work examining the economic aspects of alternative offset metrics, and none in the context of a market. Consideration of the incentives for landowners, as potential offset providers, and developers, as potential buyers of credits, is critical when considering the real-world policy implications of choosing a specific offset metric. Landowners base their decisions on whether to create offset credits on benefit:cost ratios of competing, mutually exclusive land uses. The expectation is that the least profitable land parcels are the ones most likely to be conserved, which determines the spatial location of credits Drechsler (2022). Developers base decisions on the value of different parcels for development and the expected costs of buying offsets. For both parties, the choice of the metric is likely to affect these decisions and thus the spatial distribution of biodiversity, but no work to date has explored this.

To address this gap, we developed an ecological-economic model to compare the ecological and economic outcomes of offsetting for two metrics: one based on habitat and one based on species. We compared these metrics in the specific context of an offset market in which farmers supply credits to housebuilders who are required by law to acquire sufficient credits to offset the predicted impacts of land-use change. We parameterized our model with data from a particular case study system to ensure that meaningful patterns of spatial variation were represented in the model. We aimed to improve understanding of the relationship between the ecological and economic aspects of offsetting and how the offset metric choice influences both components.

METHODS

Theoretical framework and hypotheses

We developed an ecological-economic model of a biodiversity offset market for an existing landscape. The landscape was divided into parcels, with each parcel owned by a single landowner and classified as developed or undeveloped. We assumed that undeveloped land was owned and managed by farmers and some developers wished to acquire this undeveloped land for housing development. Farmers' default land use was assumed to be for agricultural purposes, namely, crop or livestock production.

The decision-making process of agents (landowners) was simplified, and economic decisions were modeled based on the economic rent (profit) generated by each land parcel in competing uses (development, agricultural land use, or conservation land use). We compared two types of rent: agricultural rent (i.e., the difference between revenues from livestock and crops and variable costs) and potential development rent of land for housing. We assumed that for a farmer to switch from agriculture to conservation, the farmer must be offered a biodiversity offset credit value equal at minimum to the agricultural rent forgone. That is, the farmer must believe that the reduction in agricultural income on a given land parcel will be compensated for by the price they can sell the resultant offset credit for. Conversely, for a developer, the potential rent from housing development must be greater than rent under the current agricultural land use for them to choose to develop new housing. A developer must factor in the need to purchase offset credits to allow their development to proceed. We assumed agricultural and development rents varied across the landscape due to differences in land productivity for farming and house buyers' preferences over where to live.

We focused first on an offset policy that aimed to secure no net loss of a specified habitat (our approach could also be applied to a net gain policy [Simpson et al., 2021]). Developers must purchase credits equal to the number of hectares of habitat lost due to development. Farmers undertake habitat creation and restoration actions on undeveloped land to generate these offset credits. Credits are measured based on hectares of habitat created, and there is no weighting for habitat quality to support certain species. As a result, the abundance of different species may increase or decrease across the land parcels. We tested the following two hypotheses: trading habitats leads to a net gain in species if the potential development rent is negatively correlated to potential species abundance on sites that offer low agricultural rent (and are thus prone to being used for offsets of development) (hypothesis 1) and trading habitats leads to a net loss for species if the potential development rent is positively correlated to potential species abundance on sites that offer low agricultural rent (and are thus prone to being used for offsets of development) (hypothesis 2).

There is an expectation that landowners are profit maximizers; thus, we assumed that land parcels with the highest predicted development rent would be developed first and parcels with the highest agricultural rents would remain farmland. We also assumed that parcels with the lowest development rents and lowest agricultural rents would be the most likely to be candidates for offset creation. Therefore, we were interested in the correlation between development rent and species abundance on restored land parcels. A policy target that focuses solely on habitat by default can benefit species where there is a negative correlation between development rent and species abundance (Figure 1). In contrast, where there is a positive correlation between development rent and species abundance, there will be a decline in species abundance, despite no net loss of habitat.

Our second offset policy focused on no net loss in the abundance of a specified species. Under this policy, the regulator specified a conservation-oriented land management practice that was expected to benefit the species targeted by the no net loss policy. Farmers could choose to adopt this land management practice and generate offset credits, which were measured and then awarded depending on the predicted increase in abundance of the target species. Land parcels now had an ecological weighting based on their predicted ability to support the species as specified in the policy target, in contrast to the habitat metric case. The overall abundance of the target species would be maintained across the landscape after offset trades took place because the no-net-loss rule governed the rate at which development sites lost to conservation are substituted with new offset sites. However, the spatial distribution of the target species would likely change as a result of exchanging credits.

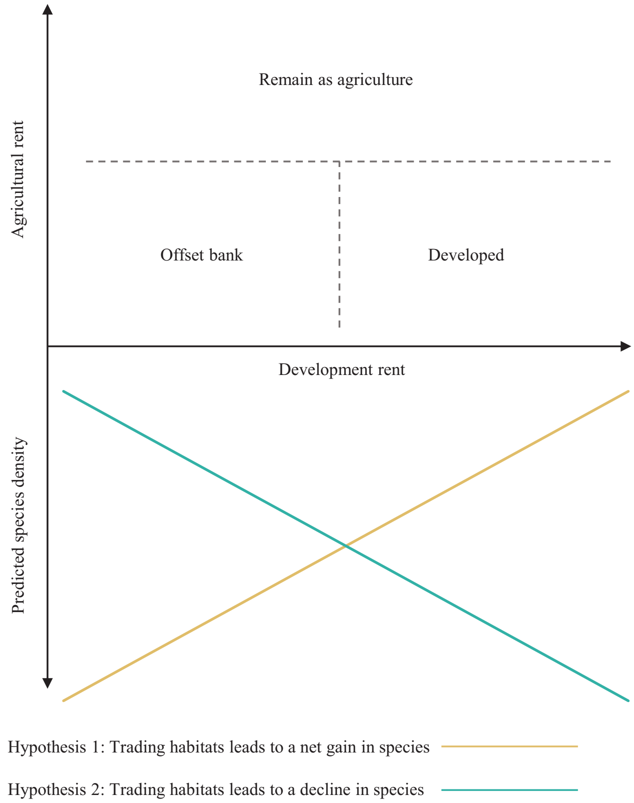

FIGURE 1 Schematic of two alternative hypotheses related to the conservation offset market. Trading habitats leads to a net gain in species if the potential development rent is negatively correlated to potential species abundance on sites that offer low agricultural rent (and are thus prone to being used for offsets of development) (hypothesis 1) and trading habitats leads to a net loss for species if the potential development rent is positively correlated to potential species abundance on sites that offer low agricultural rent (and are thus prone to being used for offsets of development) (hypothesis 2)

Case study region and offset metric

We applied our biodiversity offset model to the Inner Forth Estuary in central Scotland (Figure 2). The region is characterized by a heavily industrialized estuary surrounded by increasingly urbanized landscapes in the east, shifting toward low lying agricultural land and upland moors in the west. Alongside agricultural land, undeveloped areas contained a mosaic of biodiversity-rich areas, including seminatural grasslands subject to low-intensity use, wetlands, marshlands, and heather uplands, some of which are protected through the EU Habitats and Wildlife Birds Directive (92/43/EEC and 2009/147/EC). However, biodiversity-rich areas outside these designated sites face pressure from the growing human population. As a result, our habitat-based policy target was no net loss of low-intensity grassland. In our case study, low-intensity grassland is restored when farmers remove livestock from currently grazed grassland or cease arable cropping practices and create new grassland. Costs associated with grassland conversion from arable land are minimal, typically involving soil cultivation and seeding only (Knight & Overbeck, 2021).

In order to test our hypotheses, it was important to choose a species metric that aligned with the no net loss of low-intensity grassland policy so that we could explore whether the landscape-scale outcomes were different under the habitat and species metrics. Therefore, we compared the no net loss of low-intensity grassland metric with two species-based metrics: no net loss in the abundance of the Eurasian curlew (*Numenius arquata*) and no net loss in the abundance of the northern lapwing (*Vanellus vanellus*). Both these species depend on access to

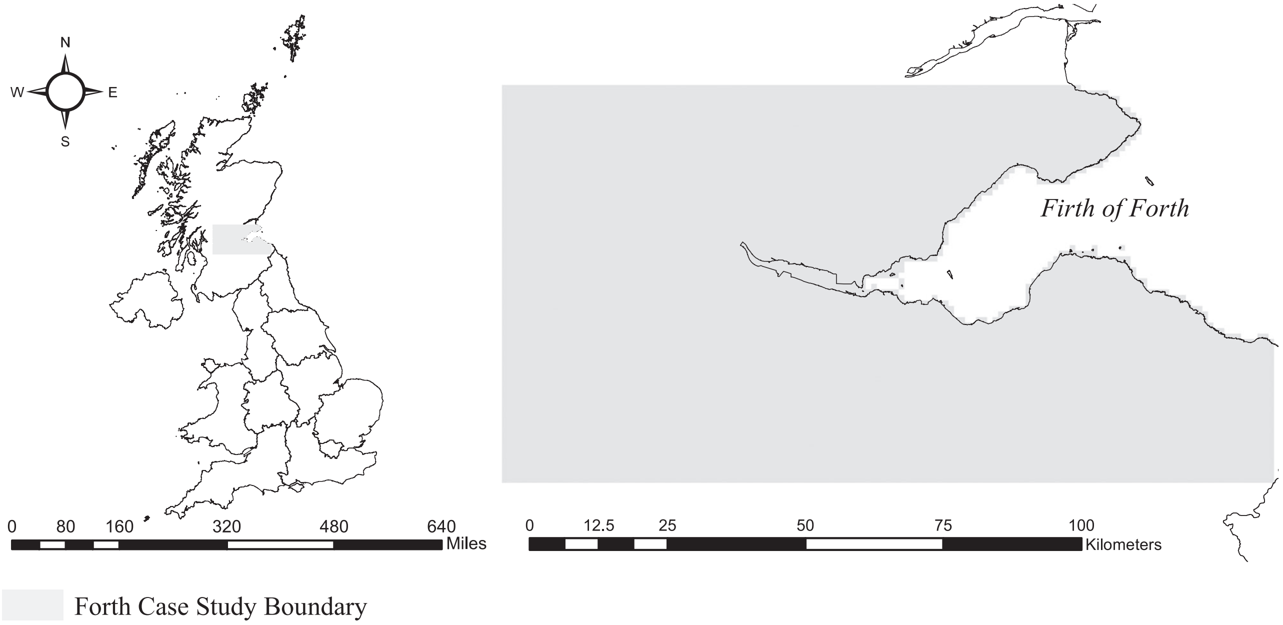

FIGURE 2 Region of the case study on the ecological-economic modeling of alternative metrics for biodiversity offsets in a market for biodiversity offsetting

suitable grassland during the breeding season; consequently, we expected that undertaking restoring low-intensity grassland on agricultural land would increase the abundance of both species, hence generating offset credits. We modeled the biodiversity offset market for each species independently so that we could explore the ecological impact on the species not defined under the no-net-loss policy.

Habitat, species, and cost data

We divided our landscape into 1-km^2 land parcels (100 ha). Each land parcel contained data from five spatially referenced data sets covering land classification, crop distribution, housing values, protected area status, and lapwing and curlew abundance and distribution. Land use was classified into 33 types, including urban, improved grassland, arable, and horticulture (Rowland et al., 2017), which allowed us to identify land parcels suitable for development and agricultural land parcels suitable for low-intensity grassland restoration.

We assumed that new housing development could not take place in designated protected areas (Figure 2) and or on certain habitat types (e.g., saltmarsh, fen, coniferous forest, broadleaf forest, and inland rock habitats). The value of undeveloped land for new housing development was calculated using Her Majesty's Land Registry transactional data combined with the existing land-use classifications (see Appendix S1 for more details). We calculated the gross margin (rent) of agricultural parcels by combining crop coverage with the associated gross margin data available in the Farm Management Handbook (Beattie, 2019).

To predict the abundance of lapwing and curlew across the landscape under the current land use, we developed species abundance models (SAMs) for lapwing and curlew (Barker et al., 2014). We also used the SAMs to identify which agricultural land parcels could offer species offset credits if the parcel were restored to low-intensity grassland (details on the SAM are given in Appendix S2).

Ecological-economic model

An agent-based model was developed in Stata MP 16 to model landowners' choices based on the relative economic returns of the alternative land management options for each parcel. The model consisted of three stages. First, the SAM predicted the current abundance lapwing and curlew across the case study region based on current land use. This provided us with a no-net-loss baseline for the target species. Second, the SAM was used to predict changes in the abundance of lapwing and curlew as a result of landowners restoring their agricultural land to low-intensity grassland. This allowed us to calculate the number of offset credits a land parcel could supply by subtracting the predicted increase in species abundance from the current species abundance. For example, a land parcel containing a mix of cereal crops supported zero lapwings. If the farmer restored the parcel to low-intensity grassland and the model predicted this parcel would support three lapwings, this generated three lapwing offset credits. The calculation of the low-intensity grassland offset credits was more straightforward because this did not require the use of the SAM. The grassland credits were calculated as the grassland cover in the parcel if the agricultural land is restored, minus the current grassland cover in a parcel in hectares. For example, if a farmer restored 90 ha of agricultural land to low-intensity grassland, this generated 90 ha of credits.

The agent-based model then determined the profitability of each land parcel under each of three mutually exclusive land-use options: development, offset provision, or current land use. By integrating this profitability with the offset requirements, potential supply or demand or both for offset credits for each land parcel was determined.

Finally, we modeled a sequential trading process based on these spatially explicit demand and supply curves and the no-net-loss policy goal. We assumed a mechanism existed in the offsets market that collects supply offers from all potential suppliers (farmers), in terms of their minimum willingness to accept (WTA) compensation for the offer of a given offset credit. We assumed the same mechanism collects demand offers from all potential buyers, in terms of their maximum willingness to pay (WTP) for each offset credit. These supply and demand offers were then ordered from highest to lowest (demand) and lowest to highest (supply). Finally, potential buyers and sellers were paired sequentially: the buyer with the highest WTP was paired with the seller with the lowest WTA. The buyer with the lowest WTP was paired with the seller with the highest WTA until no more gains from trade can be realized.

This procedure allowed us to calculate the market-clearing (equilibrium) price for offset credits. Using this equilibrium price, we then determined whether a land parcel remained under current land use, was supplied offsets, or was developed for housing. Three landscape configurations were generated using the three alternative metrics (no net loss of low-intensity grassland, no net loss of curlew, and no net loss of lapwing). Using ArcGIS, we compared where development would take place under each metric, how the distribution of low-intensity grassland would shift, and the changes in the abundance of lapwing and curlew. Based on this, we examined whether no net loss of low-intensity grassland could benefit the lapwing and curlew or whether a more targeted species metric was needed to secure the conservation of these species. Details on the agent-based model are given in Appendix S3.

RESULTS

Habitat metric

Under the no net loss of low-intensity grassland metric, there was a predicted loss of 674 lapwings and 978 curlews. Of the 409 low-intensity grassland parcels developed, 345 of these contained at least one lapwing (Figure 3) and 363 of these parcels contained at least one curlew (Appendix S3). Lapwing abundances were significantly lower (mean [SD] = 0.50 [0.57]) on restored low-intensity grassland parcels compared with lapwing abundances on the original grassland parcels (mean = 1.37 [2.25]) (t_{145} = 14.61, p = <0.001). A similar result was found for curlew (Appendix S3).

The decline in lapwing and curlew arises in part due to the heterogeneity of the bird distributions across the landscape, but is also influenced by the characteristics of the supply and demand sides of the offset market. To explore this further, we calculated pairwise correlations between the abundances of lapwing and curlew prior to offsetting, agricultural rent of a parcel, and development rent of a parcel. We calculated these pairwise correlations for the parcels that were traded under the grassland metric ($n = 508$) (Figure 4).

For both species, development rents were significantly and positively correlated with species abundance (lapwing: $r = 0.60$, $n = 508$, $p < 0.001$; curlew $r = 0.54$, $n = 508$, $p < 0.001$). As a result, there was a disproportionate conversion of low-intensity grassland habitat with high numbers of lapwing and curlew to new housing. In principle at least, gradients in agricultural rent had the potential to alter the choice of whether to develop or not (Figure 1). Potential development rent and agricultural rent (the farmland gross margin) showed a significant negative correlation ($r = -0.56$, $n = 508$, $p < 0.001$) (Figure 4); thus, the parcels with the lowest agricultural rents also aligned with the parcels most likely to be developed. Lapwing and curlew abundances were also negatively correlated with agricultural rents (lapwing: $r = -0.28$, $n = 508$, $p < 0.001$; curlew: $r = -0.42$, $n = 508$, $p <$ 0.001). Thus, agricultural parcels with the lowest rents that benefited lapwing and curlew were the same parcels that were more likely to be developed for housing than restored to grassland offsets. Agricultural parcels that benefited curlew and lapwing were more likely to be developed than restored to a grassland offset.

Our results confirmed our hypothesis that trading habitats leads to a net loss for species if the potential development rent is positively correlated to potential species abundance on sites that offer lower agricultural rent (and are thus prone to being used for offsets of development).

Species metrics

The amount and location of new housing development on low-intensity grassland were broadly similar for the lapwing species metric (Figure 5) and curlew species metric (Figure 6). Development took place on grassland parcels with low abundances of the target species. For the lapwing metric, the mean number of lapwings lost to development per grassland parcel was 0.54. For the curlew metric, the mean number of curlews lost to development per grassland parcel was 0.37. For both species, their respective offset sites were located near the coastal margin and upland regions: both areas where predicted abundance for lapwing and curlew was high. There was a significant difference in lapwing abundance between the parcels that became offset supply sites (mean [SD] = 4.71 [8.57]) and those that were either developed or remained in the original land use (mean = 1.59 [4.12]) (t_{8347} = 7.82, $p \leq 0.001$). There was also significant difference in curlew abundance between the parcels that became offset supply sites (mean = 3.62 [5.93]) and those that were either developed or remained in the original land use (mean = 1.22 [2.25]) (t_{8347} = 8.83, p = <0.001).

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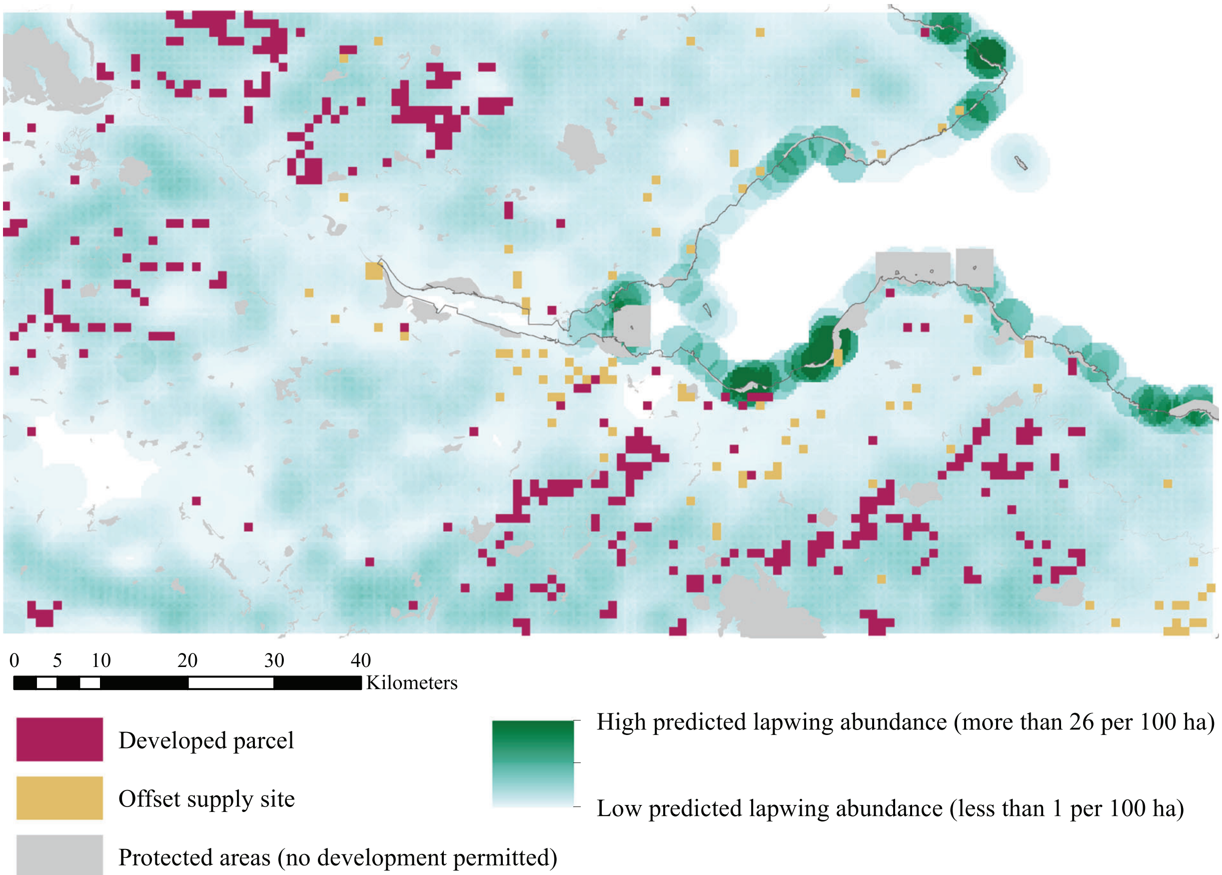

FIGURE 3 The location of agricultural land parcels restored to low-intensity grassland and the location of low-intensity grassland parcels converted to housing development under a biodiversity offset market that uses a metric based on the no net loss of low-intensity grassland. The predicted abundance of lapwing per land parcel under the original land use is included to highlight the impacts of development on lapwing

Comparison of habitat and species metrics

The landscape-scale outcomes were substantially different depending on the choice of either a habitat- or species-based metric (Table 1). The distributions of curlew and lapwing abundance were heterogeneous across grassland parcels throughout the landscape; thus, there was divergence in grassland parcels that were traded under the habitat and species metrics. If the spatial distribution of lapwing and curlew abundances were homogenous, we expected the same parcels to have been traded, regardless of the metric chosen (details on this finding are given in Appendix S2).

Significantly more low-intensity grassland parcels were developed for housing under the lapwing species metric (mean [SD] = 1.96 [9.12]) compared with the grassland metric (mean = 0.54 [3.55]) (t_{16696} = 13.27, p = <0.001). Despite higher levels of development under the lapwing species metric, there were fewer grassland offsets created. The increases in grassland under the habitat metric (mean = 0.54 [5.8]) were significantly greater than gains in grassland under the lapwing metric (mean = 0.29 [3.16]) (t_{16696} = 3.48, p <0.001). Consequently, there was a substantial loss of grassland under the lapwing species metric (16,267 ha). This finding was the same as for the curlew metric, for which offset trading resulted in a loss of 19,045 ha of grassland.

DISCUSSION

Using an ecological-economic modeling framework, we simulated a biodiversity offset market that secured no net loss of three alternative metrics: no net loss of low-intensity grassland (habitat-based), no net loss of lapwing (species based), and no net loss of curlew (species based) for a case study region. For each of these metrics, there were significant off-market impacts on the related habitats and species that were not explicitly protected by the no-net-loss policy.

Our results showed that none of the three metrics adequately captured the indirect benefits of offsetting on related habitats or species. There were substantial declines in lapwing (loss of 678) and curlew (loss of 964) under the no net loss of low-intensity grassland metric. This is in contrast to the wider literature (Franks et al. [2018] contains a summary) and highlights that curlew and lapwings benefit from restoration of low-intensity grassland. Furthermore, under the species-based offset metrics, there were also declines in the nontarget

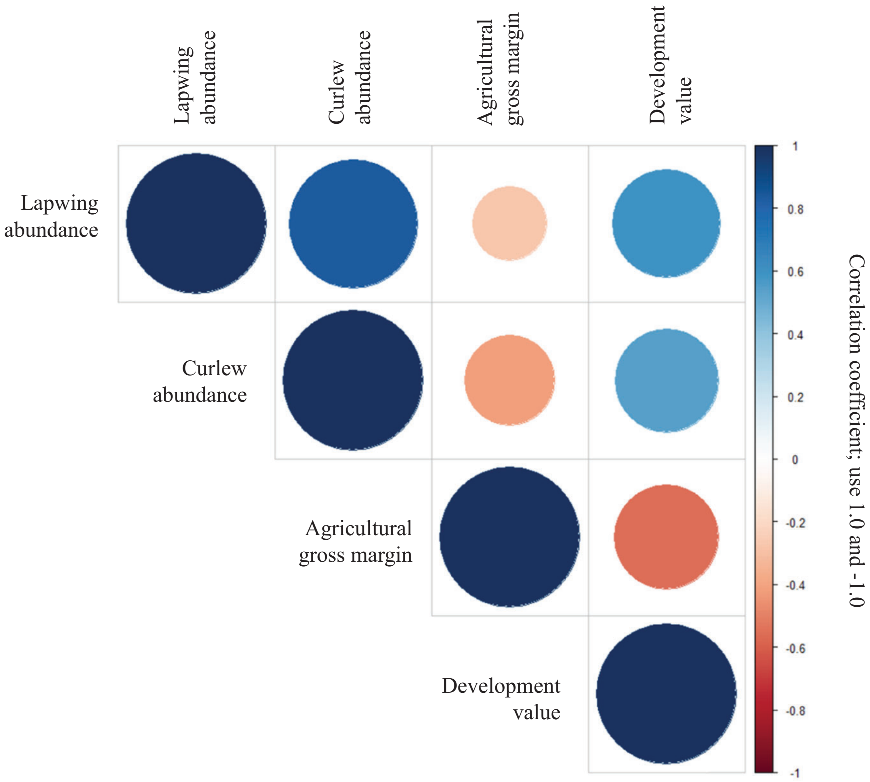

FIGURE 4 Pairwise correlation matrix for current abundances of curlew and lapwing, agricultural gross margin, and potential development value of land (blues, positive correlations; red, negative correlations; color intensity and size of circle proportional to the correlation coefficients)

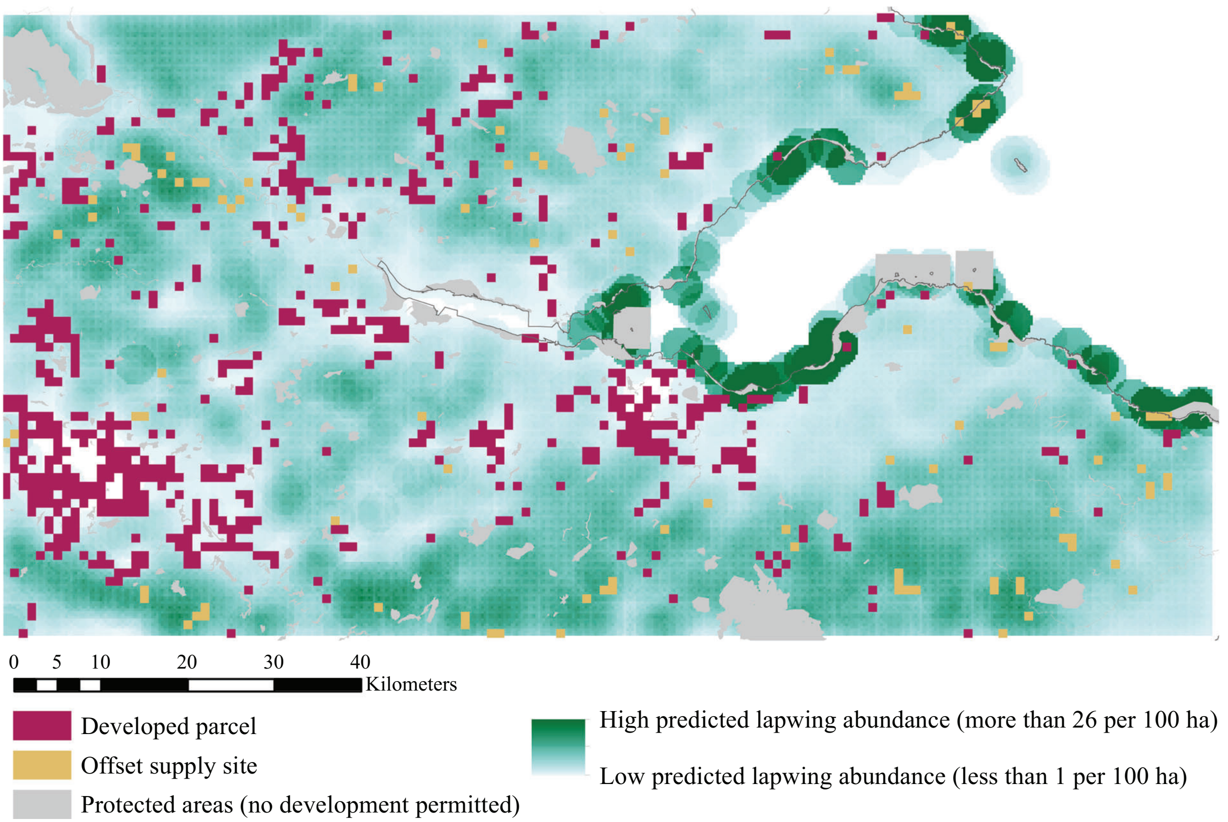

FIGURE 5 The location of agricultural land parcels restored to low-intensity grassland and the location of low-intensity grassland parcels converted to housing development under a biodiversity offset market that uses a metric based on no net loss in abundance of lapwing

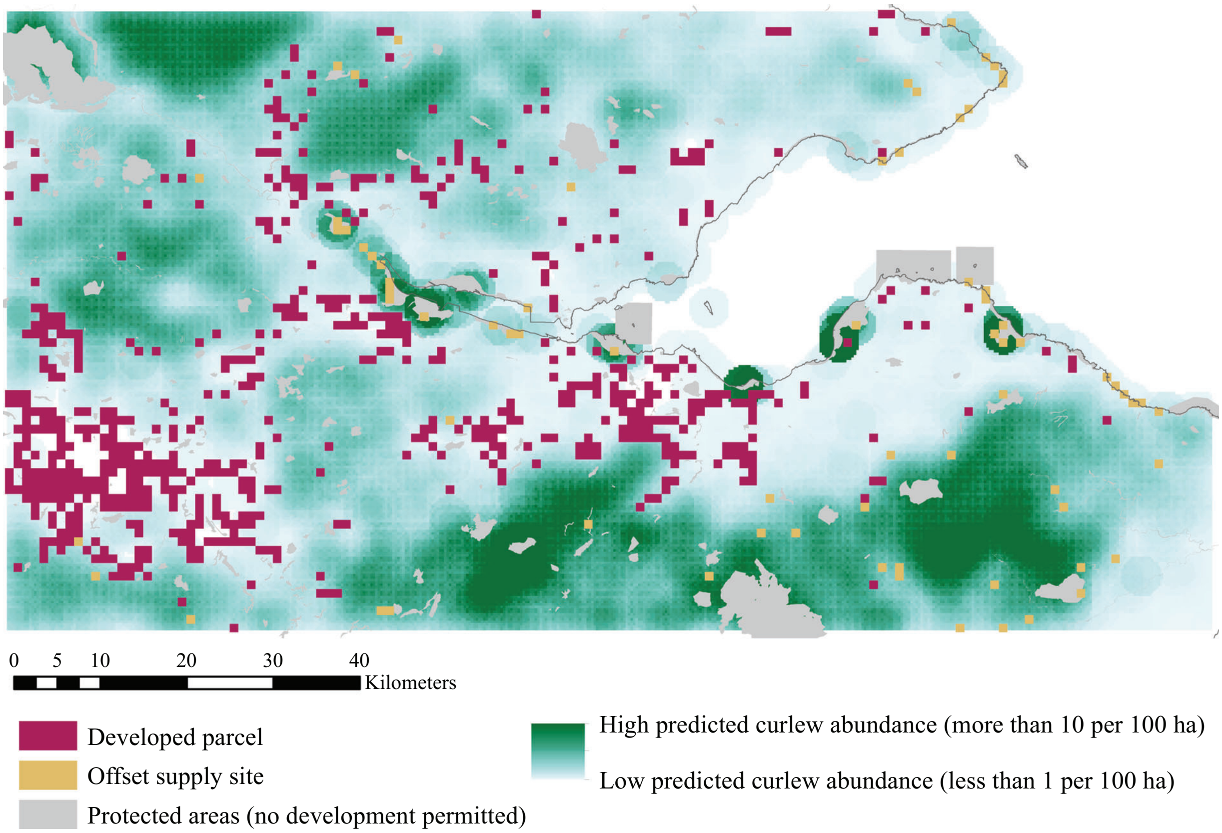

FIGURE 6 The location of agricultural land parcels restored to low-intensity grassland and the location of low-intensity grassland parcels converted to housing development under a biodiversity offset market that uses a metric based on no net loss in abundance of curlew

TABLE 1 A comparison of losses and gains in grassland and the abundance of lapwing and curlew under alternative offset metrics for the simulated biodiversity offset market in the case study region

	Grassland metric	Lapwing metric	Curlew metric
Grassland (ha) lost to development	4554	16,436	19,405
Grassland (ha) restored	4536	169	76
Lapwings lost to development on grassland	674	169	231
Predicted lapwings on restored grassland	0	169	50
Curlews lost to development	978	192	75
Predicted curlews on restored grassland	14	50	76

species (although not to as large an extent as under the grassland metric). There was a net loss of 181 lapwings under the curlew metric and a net loss of 142 curlews under the lapwing metric.

The decline in lapwing and curlew under the grassland metric was related to the economic choices faced by landowners. For a landowner to choose to become an offset supplier, offset supply must be more profitable than the current land use. The expectation is, therefore, that the least profitable land parcels are the ones most likely to be conserved (Drechsler, 2022). We found that for lapwing and curlew, there was a significant positive correlation between the predicted species abundance and the most profitable parcels for future development. Thus, if a metric does not specify no net loss of either species, there will be a significant loss in these species due to development. Moreover, development rent and agricultural rents were significantly negatively correlated and predicted species abundances were also negatively correlated with higher agricultural rents. Thus, agricultural parcels that benefit curlew and lapwing were more likely to be developed than restored to a grassland offset and parcels restored to create new grassland offset sites were unlikely to significantly benefit curlew or lapwing. This would not necessarily hold in other landscapes or for different metrics. Indeed, the opposite result is possible if the policy target is no net loss of habitat because other plant and animal species may increase. We would expect to find this outcome where there is a negative correlation between species abundance and expected development rents on sites that offer lower agricultural rent (and are thus prone to being used for offsets of development). In such a situation, habitat-based metrics would secure additional ecological gains and meet the policy community's aim to have a simpler metric that can capture indirect ecological benefits. However, relying on a habitat-based metric to secure no net loss of a specific species is rarely successful (Cristescu et al., 2013; Marshall et al., 2020b; Quétier et al., 2014).

In contrast to the habitat-based metric, the species metric can be viewed more positively. The two species-targeted offset markets resulted in outcomes in which the highest value ecological sites were protected; no development took place on low-intensity grassland parcels that contained more than two lapwings or curlew. On the supply side, as expected, market-derived incentives encouraged grassland restoration on agricultural parcels that offered the greatest increases in lapwing and curlew at the lowest opportunity cost, but they also pushed offset supply to focus on a few high-value grassland sites in areas with already high numbers of curlew and lapwing. A consequence of this was a significant decline in grassland area under both species-based metrics. A natural question to ask then is: Is a large amount of habitat loss elsewhere what policy makers intended or what the general public wants? From a societal perspective, this would result in a loss of easily accessible greenspace and could have a significant impact on the well-being of local communities (Griffiths et al., 2019; Jones et al., 2019).

A further consideration for the species metric is the interplay between the economic and ecological models. The economic model was designed to identify parcels that offer the most offsets at the lowest cost (which it achieved). However, this highlights the potential limitations in the underpinning ecological models. Species abundance predictions were less reliable for land parcels in areas in our region where data were sparse and for the few parcels that hold particularly high abundances of birds. Given that the economic model focuses on identifying the smallest number of sites that can ensure no net loss in abundances, the economic model will inevitably identify land parcels for which the uncertainty in our predicted species abundances from the ecological models is highest.

We recognize there are several limitations to our modeling approach. From an ecological perspective, our model does not take into account temporal dynamics because we included no time lags between losing an ecologically valuable land parcel to development and the offset site being created. This is equivalent to assuming that the offset bank will only sell credits where and when the predicted ecological benefit has already been realized. A dynamic model exploring ecological and economic time scales would offer an interesting extension. There is also a need to expand the framework to consider additional habitat types that qualify as offsets beyond grassland and to include the restoration cost data associated with these habitat types. We designed our offset market for an existing U.K. landscape, but this approach could be replicated for other areas worldwide to facilitate comparison of the landscape-scale impacts of different offset metrics for a trading scheme. The work could also be expanded to take into account multiple environmental outcomes (rather than just changes in habitats and species) or a broader range of biodiversity indicators (subject to data availability).

From a policy perspective, each of the metrics we considered achieve its intended policy target: no net loss of grassland, no net loss of curlew, or no net loss of lapwing. However, the underlying species distributions, layered with the agricultural and development rental values of parcels, resulted in very different landscape outcomes, depending on the metric chosen. What these results show is that if the policy maker is aiming for the metric to act as an indicator to mitigate impacts on a range of closely related habitats and species, then a simple no net loss target is not adequate. If policy makers wish to secure multiple outcomes from an offset policy, then these must be established within the policy target. Choosing to focus on a single indicator species will not deliver multiple target outcomes for biodiversity (Armsworth et al., 2012). The simpler (theoretical) solution to this is to specify these multiple outcomes within the policy, that is no net loss of grassland and no net loss of lapwing. However, with the focus on biodiversity offsetting moving toward securing ecosystem service benefits, such as recreation and reduced flood risk, this would require a highly complex policy prescription and a much more complex offset metric. Moreover, more complex offset metrics increase the costs of implementing the scheme and are likely to reduce the number of trades and hence the economic efficiency of this policy instrument (Needham et al., 2019).

Rather than developing a complex offset credit, an alternative would be to offer an additional prescription within the no net loss policies for the habitat or species metrics. For the habitat metric, the policy prescription would include a focus on increasing the quality of the restored parcels in terms of ecological productivity. One way to achieve this would be to differentiate grassland parcels based on the habitat quality condition assessments. For the species metric, we would be looking to increase the number of grassland parcels restored across the landscape. To encourage a greater number of offset sites, there could be a limit on the number of species credits that could be sold for a single parcel (thus stimulating additional landowners to choose to supply offsets). This has two advantages. First, it overcomes the problems identified in the ecological-economic modeling framework in which the economic model presses on the upper tail of the predictive ecological model and selects the offset sites with very high predicted species abundance. Second, by increasing the number of offset sites, it reduces the social impacts associated with large losses in accessible grassland.

However, under each of these additional policy prescriptions, the impact on the functioning of the offset market itself would need to be taken into account if the ultimate goal is to facilitate offset trading to enable development and conservation priorities to be jointly met. For example, as Simpson et al. (2021) show, increasing a net gain requirement on developers results in fewer landowners choosing to supply offsets and thus less land converted to conservation.

Our model showed that there are significant economic and ecological implications of the choice of metric for a biodiversity offset trading scheme. Because these differences in outcomes relate to predictable spatial relationships in observable variables (agricultural profits and development rents), our results have broad implications for biodiversity offset schemes globally. It is clear that, if one wishes to secure the most ecologically beneficial design of offsets policy, whether that is based on habitats, species, or some other metric, one needs to understand the economic decision-making processes of the landowners. One also needs to design incentive-based policies that offer the highest incentives for conserving and enhancing the most ecologically beneficial sites in a landscape.

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

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