Ecological and Economic Implications of Alternative Metrics in Biodiversity Offset Markets

Abstract

Consideration of the incentives facing both landowners as potential biodiversity offset providers, and developers as potential buyers of credits, is critical when considering the real-world policy implications of choosing a specific offset metric and the resultant impacts on biodiversity. The expectation is that the least profitable land parcels are the ones most likely to be conserved, which 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 defined under the no net loss policy. We simulated a biodiversity offset market for a case study landscape, linking species distribution modelling and an economic model of landowner choice based on economic returns of the alternative land management options (restore, develop, or maintain existing land use). The biodiversity offset markets for the habitat and species metrics achieved no net loss of the intended metric. However, the underlying species distributions, layered with the agricultural and development rental values of parcels, resulted in very different landscape outcomes depending on. Neither metric adequately captured the indirect benefits of offsetting on related habitats or species. Where policymakers 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, if we wish to secure the most ecologically beneficial design of offsets policy, we need to understand the economic decision-making processes of the landowners.

Keywords:

Biodiversity metrics. No net loss. Biodiversity loss. Simulation model.

Article Impact Statement:

Design policies that offer the highest incentives for conserving and enhancing the most ecologically beneficial sites in a landscape.

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 which allow us to reconcile development pressures with biodiversity conservation. Biodiversity offsets are one such policy option that is becoming increasingly applied to respond to these pressures (Moilanen & Kotiaho 2020).

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, restore) have been undertaken (Alridge et al 2019). 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 towards Net Positive Impact and Biodiversity Net Gain, which aim to improve the state of the environment compared to the pre-development state (Bull & Brownlie 2017; Moilanen & Kotiaho 2020; McVittie & Faccioli 2020).

In this paper, we focus 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 2020). Landowners can choose to manage land for conservation, generating offset credits which can then be sold to a developer who is required to mitigate development impacts, for example from house building, on some measure of biodiversity. By establishing an appropriate rate of exchange between sellers and buyers, markets can, in theory, achieve 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 (Heal 2005; Bull et al 2013). From an economic perspective, markets require goods to be grouped into simple, measurable, standardized units to foster exchangeability and market efficiency (Salzman and 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 widely used offset metrics use a combination of habitat area, vegetation, and site condition scores (Parkes 2003; 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, non-target 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 2019).

Recent literature has begun to assess alternative offset metrics that include more detailed species data and compare their performance with habitat-based metrics (Maseyk et al 2016; McVittie and Faccioli 2020; Marshall et al 2020b). However, there has been little quantitative work examining the economic aspects of alternative offset metrics, and none within the context of a market. Consideration of the incentives facing both 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 over 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, 2021). Developer’s 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 impact these decisions, and thus on 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 alternative metrics: one based on habitat and one based on species. We compared these two metrics in the specific context of an offset market where 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 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 offset metric choice influences both components.

Methods

Theoretical framework and hypotheses

We developed a biodiversity offset market for an existing landscape but used a simplified decision-making process. 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 currently 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.

Economic decisions were modelled based on the economic rent (profit) generated by each land parcel in competing uses (development, agricultural land use, or conservation land use). We were interested in comparing two types of rent: agricultural rent (defined as the difference between revenues from crops/livestock 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. In addition, a developer must factor in the need to purchase offset credits to allow their development to proceed. We expected agricultural and development rents to vary across the landscape due to differences in land productivity for farming and in 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, see 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, with 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:

Hypothesis 1: *Trading habitats will lead to a net gain in species if the potential development rent is negatively correlated to potential species abundance on sites that offer lower agricultural rent (and are thus prone to being used for offsets of development).*

Hypothesis 2: *Trading habitats will lead 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).*

There is an expectation that landowners are profit maximisers and such we expect that land parcels with the highest predicted development rent will be developed first, and parcels that offer the highest agricultural rents will remain farmland. Parcels with the lowest development rents and lowest agricultural rents are more likely to be candidates for offset creation. Therefore, what we are interested in is the correlation between development rent and species abundance on restored land parcels. A policy target that focuses solely on habitat by default can benefits 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 specifies a conservation-oriented land management practice that is expected to benefit the species targeted by the no net loss policy. Farmers can choose to adopt this land management practice and generate offset credits, which are measured and then awarded depending on the predicted increase in abundance of the target species. Land parcels now have 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 will be maintained across the landscape after offset trades take place since the no net loss rule governs the rate at which development sites “lost” to conservation are substituted with “new” offset sites. However, the spatial distribution of the target species is likely to change as a result of exchanging credits.

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 towards low lying agricultural land and upland moors in the west. Alongside agricultural land, undeveloped areas contain a mosaic of biodiversity-rich habitats including semi-natural grasslands that are subject only 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 out with these designated sites face pressure from the growing population requiring new housing. As a result, our habitat-based policy target is no net loss of low-intensity grassland. Low-intensity grassland is restored in our case study by farmers removing livestock from currently grazed grassland or ceasing arable cropping practices and creating new grassland. Costs associated with grassland conversion from arable land are minimal, typically involving soil cultivation and seeding only.

To enable us 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 of these species depend on access to suitable grassland during the breeding season and 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 1km^2 land parcels (100 ha); each land parcel contains data from five spatially referenced datasets covering land classification, crop distribution, housing values and protected area status, as well as lapwing and curlew abundance and distribution. Land use was classified into 33 types including (Rowland et al 2015) and this 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 within designated protected areas (indicated in Figure 2) and also on certain habitat types (such as saltmarsh, fen, coniferous forest, broadleaf forest, and inland rock habitats). The value of undeveloped land for new housing development was calculated using HM Land Registry transactional data combined with the existing land use classifications (see Appendix A 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).

We developed Species Abundance Models (SAMs) for lapwing and curlew to allow us to predict the abundance of lapwing and curlew across the landscape under the current land use (Barker et al 2014). We also used the SAMs to identify which agricultural land parcels could offer species offset credits if the parcel was restored to low-intensity grassland (see Appendix B for more details on the SAM).

The Ecological-Economic Model

An agent-based model was developed in Stata MP (Version 16) to model landowners' choices based on the relative economic returns of the alternative land management options for each parcel. The model calculated the profitability of each land parcel for housing development, offset provision and current land use. The model also identified the number of offset credits a parcel could supply if restored to low-intensity grassland, and the number of offset credits required if the parcel were developed for new housing. By integrating the profitability of the parcel with the potential offset demand and supply, we were able to construct spatially explicit supply and demand curves for offset credits. This allowed us to calculate the market-clearing (equilibrium) price for one offset credit. Using this equilibrium price, we could then determine whether a land parcel remains under current land use, supplied offsets or was developed for housing. Three landscape configurations were generated using the three, alternative metrics. 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. For more details on the design of the Agent-Based Model see Appendix C.

Results

Habitat metric

The biodiversity offset market secured no net loss of low-intensity grassland under the habitat metric: 4536 ha of grassland were to development and 4553 ha of grassland was restored. However, 345 of the 409 land parcels developed contained at least one lapwing (Figure 3). This result by itself is not concerning: all grassland parcels which are developed are offset. However, lapwing abundances were found to be significantly lower (M = 0.50, SD = 0.57) on restored low intensity grassland parcels compared lapwing abundances on the original grassland parcels (M =1.37, SD = 2.25) ($t(145)$ = 14.61, p =<0.001)). This resulted in a net loss of 674 lapwings. A similar result was found for curlew (see Appendix D).

The decline in lapwing and curlew arises in part due to the non-uniformity of the bird populations across the landscape but is also influenced by the characteristics on the demand side of the offset market. We found that lapwing abundance and development rent were moderately positively correlated (r=0.08, n = 8347, p < 0.001) (Figure 4). As a result, there was a disproportionate conversion of low-intensity grassland habitat with high numbers of lapwing to new housing. As shown in Figure 1, in principle at least, gradients in agricultural rent have the potential to alter the choice whether to develop or not. However, for our case study, the result in Figure 4 is what drives our findings because development rents tend to be substantially higher than agricultural rents (see Appendix D for the correlation between lapwing abundance and agricultural rent). Consequently, our results confirmed Hypothesis 2: trading habitats has led to a net loss in species with high development rents correlated with lapwing abundance.

Species metrics

The two simulated species biodiversity offset market secured no net loss of the metric (no net loss of lapwing under the lapwing metric, and no net loss of curlew under the curlew metric). 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 curlew 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 (M =4.71, SD = 8.57) and those that were either developed or remained in the original land use (M =1.59, SD = 4.12) ($t(8347) = 7.82$, p =<0.001)). There was also significant difference in curlew abundance between the parcels that became offset supply sites (M =3.62, SD = 5.93) and those that were either developed or remained in the original land use (M =1.22, SD = 2.25) ($t(8347) = 8.83$, p =<0.001)).

A 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. The distribution of curlew and lapwing abundance was non-uniform across grassland parcels throughout the landscape and as a result, there was divergence in grassland parcels that are traded under the habitat and species metrics. To confirm this result, we compared the equivalent costs of grassland credits under a uniform distribution of species across grassland parcels (with a focus on no net loss of grassland) (Table 1). Under the grassland metric, a grassland offset cost £499 per ha. The equivalent cost for this grassland offset using a uniform lapwing distribution was £13,588 per bird, compared to £14,127 per lapwing using the non-uniform distribution. For curlew, the equivalent cost for this grassland under the uniform curlew model was £19,683 per bird, compared to £22,005 per curlew under the non-uniformly distributed curlew model (more detail is provided on the calculation of this in Appendix D).

Consequently, significantly more low intensity grassland parcels were developed for housing under the lapwing species metric (M = 1.96, SD = 9.12) compared to the grassland metric (M =0.54, SD = 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 (M = 0.54, SD = 5.8) were significantly greater than gains in grassland under the lapwing metric (M = 0.29, SD = 3.16) (t(16696) = 3.48, p <0.001). Consequently, there is a substantial loss of grassland under the lapwing species metric (16,267 ha).

Discussion and conclusions

Using an ecological-economic modelling 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 lapwing (species based) and no net loss of curlew (species based) for a case study region. The results revealed that for each of these metrics, no net loss of the target was secured but that there were significant off-market ecological impacts.

The results show 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 despite the ecological model (and wider literature see Franks et al 2018 for a summary) highlighting that curlew and lapwings benefit from restoration of low-intensity grassland. To put these declines in context, the most recent population estimate for lapwing in Scotland is 71,500-105,600 pairs and 58,800 curlew pairs (Foster et al 2013). Furthermore, under the species-based offset metrics, there were also declines in the non-target 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 is related to the economic choices faced by landowners. For a landowner to choose to become an offset supplier, this 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, 2021). For our case study we found that for lapwing and curlew, there is a positive correlation between the predicted species abundance, the most profitable parcels for agriculture and future development rents. As a result, parcels which are restored to create new grassland-metric offset sites are unlikely to benefit curlew or lapwing. This result will not necessarily hold in other landscapes, or for different metrics. Indeed, the opposite result is possible where focusing on a simple no net loss of habitat policy target may result in increases in other plant and animal species. 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 anticipations that a simpler metric can capture indirect ecological benefits. However, previous work has shown that relying on a habitat-based metric to secure no net loss in a specific species is rarely successful (Cristescu et al 2013; Quétier, Regnery and Levrel 2014; Marshall et al 2020b).

In contrast to the habitat-based metric, the species metric can be viewed more positively. The two species-targeted offset markets resulted in outcomes where the highest value ecological sites were protected, with no development taking 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 to take place on agricultural parcels which offered the greatest increases in lapwing and curlew at the lowest opportunity cost but 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 would then be: is a large amount of habitat loss elsewhere what policymakers intended, or what the general public want? From a societal perspective, this would result in a loss of easily accessible greenspace and could have a significant impact on the wellbeing of local communities (Gordon-Jones et al 2019; Griffiths et al 2019).

A further consideration for the species metric is the interplay between the economic and ecological models. The economic model is designed to identify parcels that offer the most offsets at the lowest cost (which it has achieved). However, this highlights the potential limitations in the underpinning ecological models, which are less reliable for land parcels in areas in our region where data are sparse, or 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 that there are several limitations to our modelling approach. From an ecological perspective, the modelling does not take into account temporal dynamics as we include 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 which qualify as offsets beyond grassland and to include the restoration cost data associated with these habitat types. We have designed our offset market for an existing UK landscape, but this approach could be replicated for other areas worldwide looking to compare 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 considered here achieves their intended policy targets: no net loss of grassland, no net loss of curlew, or no net loss of lapwing. However, we have shown that the underlying species distributions, layered with the agricultural and development rental values of parcels, result in very different landscape outcomes depending on the metric chosen. What these results show is that if the policymaker 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. Our reason for exploring a single policy outcome in this paper was the simplicity in the trading rules by allowing us to trade like for like. That is, the more complex the units of exchange, the more difficult it is to establish a market where trades take place. What we show, however, is that if policymakers 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, i.e., no net loss of grassland *and* no net loss of lapwing. However, with the focus on biodiversity offsetting moving towards 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 might be to offer a policy fix on either the habitat or species metric. For the habitat metric, the policy fix would focus on increasing the quality of the restored sites in terms of ecological productivity. One way to achieve this would be to differentiate grassland parcels based on the habitat based on quality condition assessments. This approach falls under the broad "Extent x Condition" currencies, with the Australian State of Victoria's Habitat Hectares approach and the UK Defra Biodiversity Metric 3.0 being examples of this. 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 a single parcel can sell (stimulating additional parcels to enter the market). This has two advantages. Firstly, it overcomes the problems identified within the ecological-economic modelling framework with the economic model pressing on the fat tail of the predictive ecological modelling. Secondly, by increasing the number of offset sites it reduces large scale clearance and the social impacts associated with large losses in accessible grassland.

However, under each of these policy fixes, 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 shown in Simpson et al (2021) increasing a net gain requirement on developers results in fewer landowners choosing to supply offsets, with less land converted to conservation.

In conclusion, our modelling shows that there are significant economic and ecological implications following the choice of metric for a biodiversity offset trading scheme. Since these differences in outcomes relate to predictable spatial relationships in observable variables (agricultural profits and development rents), the results have broad implications for biodiversity offset schemes globally. It is clear that, if we wish to secure the most ecologically beneficial design of offsets policy, whether that is based on habitats, species or some other metric, we need to understand the economic decision-making processes of the landowners. We also need 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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Table 1: A comparison of offset losses and gains for the original grassland model and the uniformly distributed lapwing and curlew models.

	Grassland metric	Lapwing metric	Lapwing scaled uniformly across grassland parcels	Curlew metric	Curlew scaled uniformly across grassland parcels
Market clear price	£499 (per ha)	£14,127 (per bird)	£13,588 (per bird)	£22,005.38 (per bird)	£19,683 (per bird)
Grassland (ha) lost to development	4,554	16,436	4,554	19,405	4,554
Grassland (ha) restored	4,536	169	4,608	76	4,608

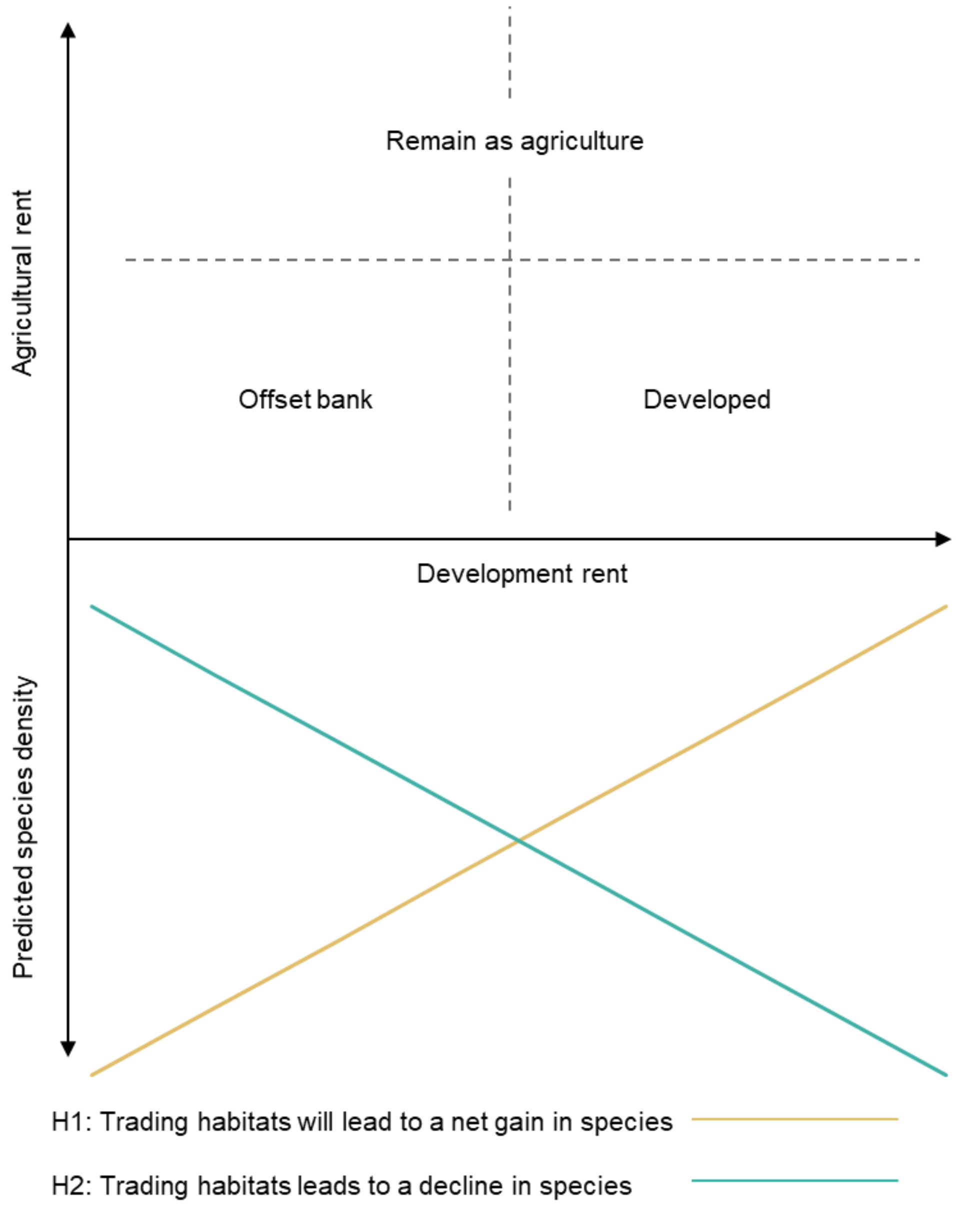

Figure 1: Schematic of the two alternative hypotheses for the offset market

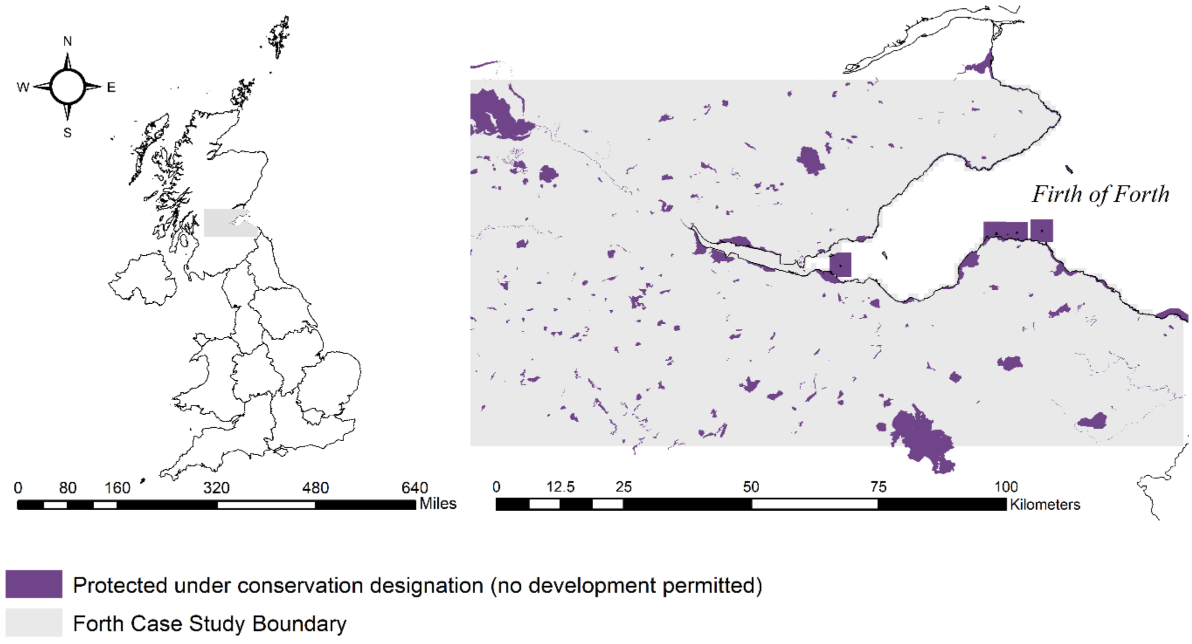

Figure 2: Overview of the case study region. Protected areas (SSSI, SPA, SAC, NNR and LNR) are protected from future housing developments.

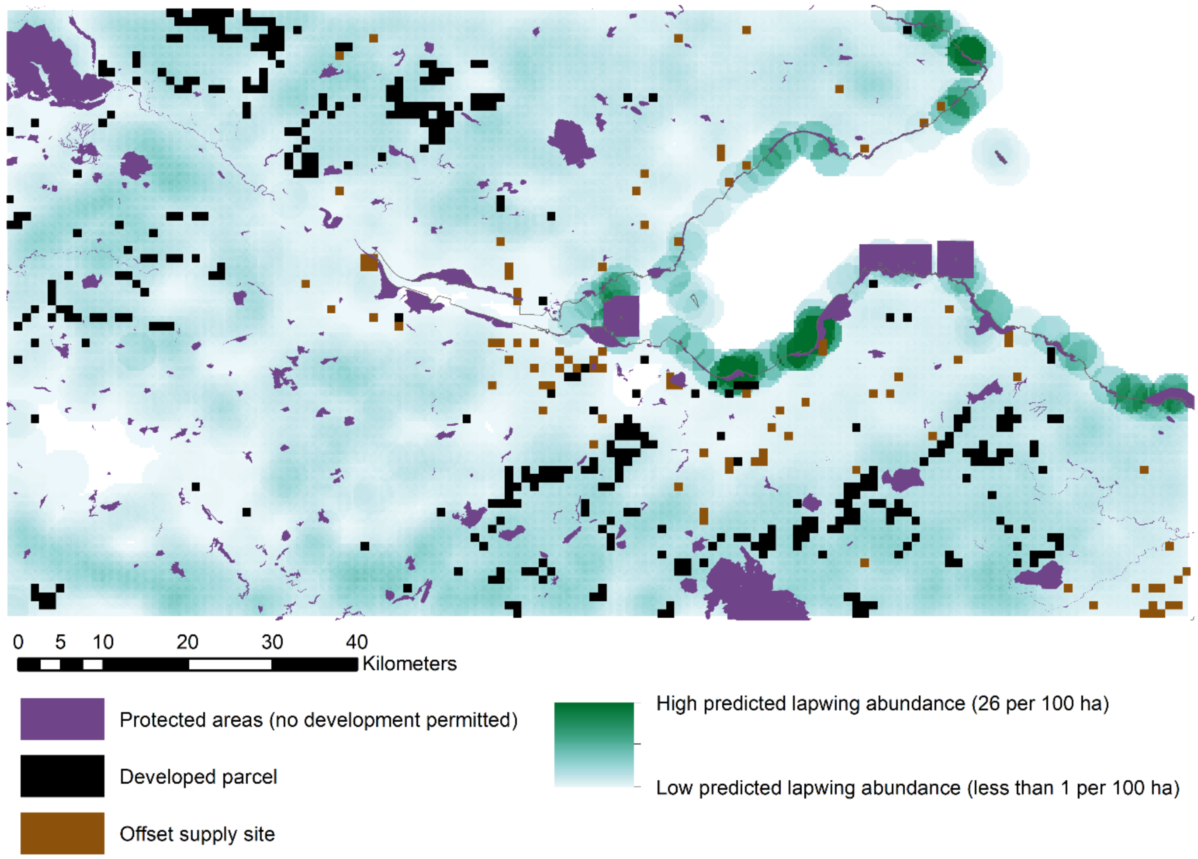

Figure 3: Landscape-scale outcomes under the grassland habitat metric

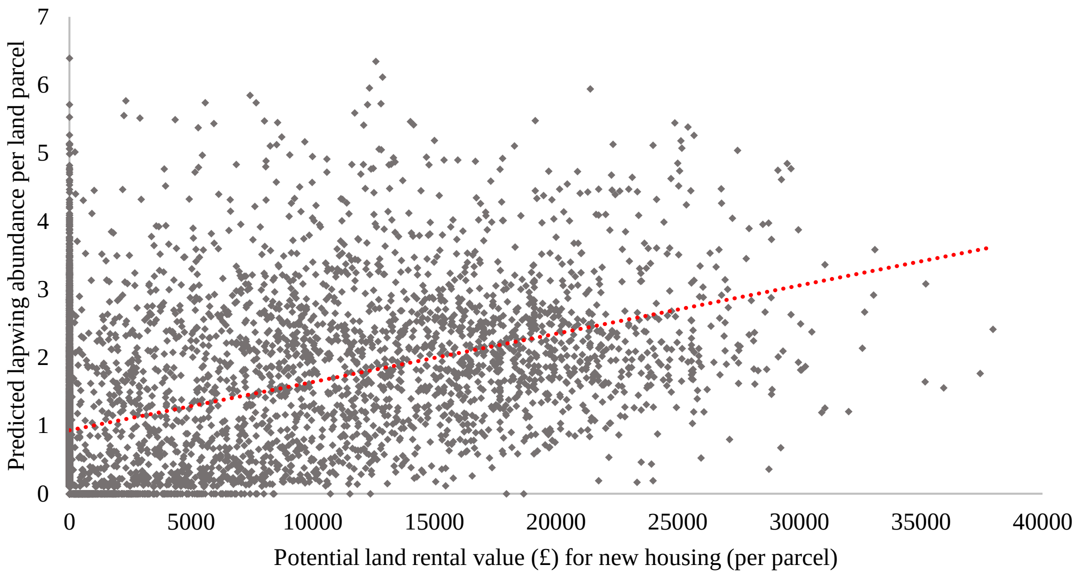

Figure 4: A comparison of the predicted number of lapwings and the potential land rental value of a parcel for housing (r=0.08, n = 8347, p < 0.001) Note: Only parcels with a positive development value that contain grassland are included in this Figure.

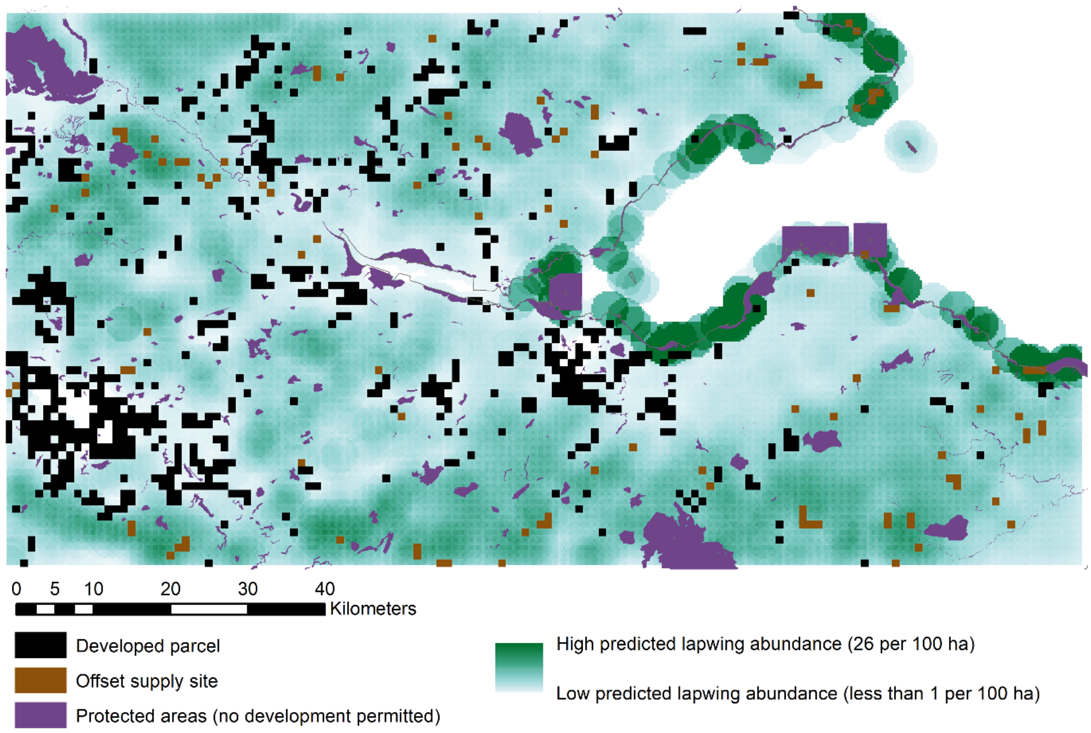

Figure 5: Landscape-scale outcomes under the lapwing species metric

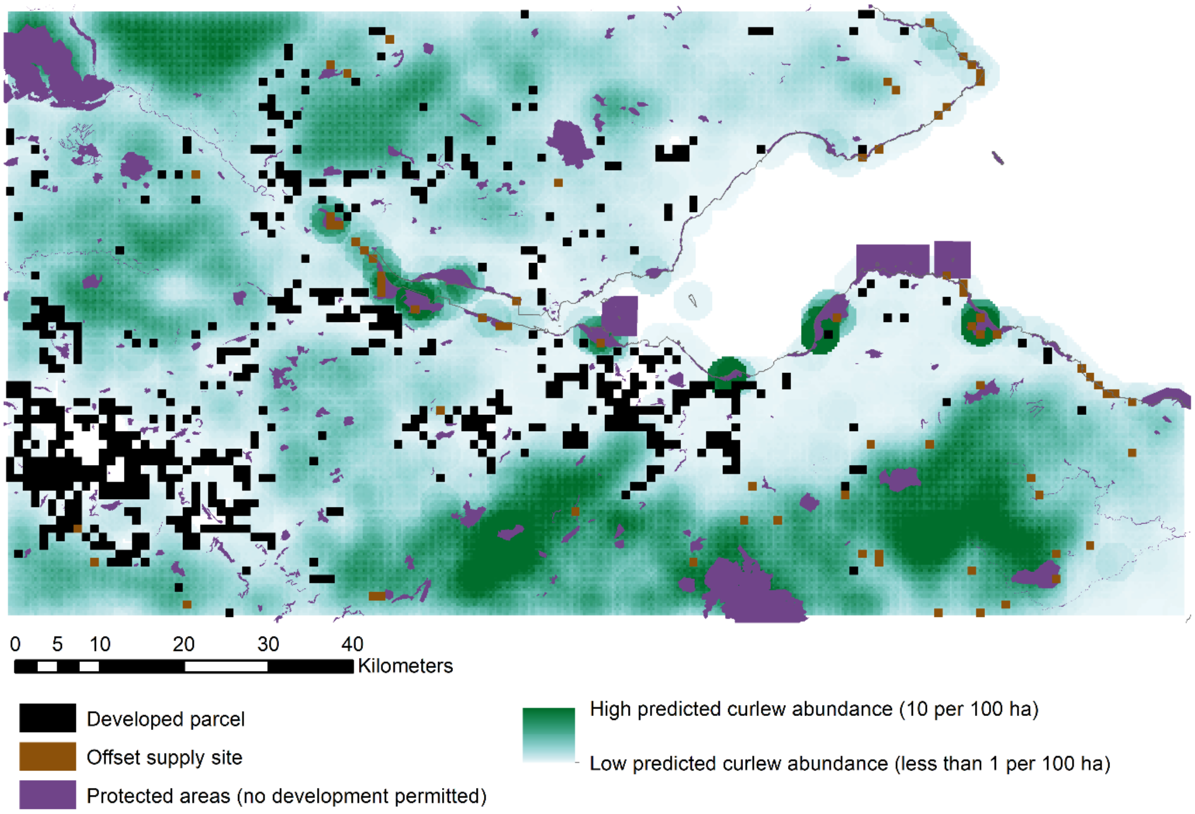

Figure 6: Landscape-scale outcomes under the curlew species metric