



Rewilding for biodiversity offsets: A case study of passive ecological restoration on lowland agricultural land for Biodiversity Net Gain in England

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ARTICLE INFO

Keywords:

Agricultural land
Biodiversity Net Gain
Biodiversity offsets
Environmental policy
Land use change
Restoration
Rewilding

ABSTRACT

England is a country with ambitious targets for habitat restoration and increased woodland cover, along with new Biodiversity Net Gain (BNG) regulations requiring most new development projects to increase overall biodiversity by 10 % (measured via the statutory Defra Biodiversity Metric). Typically this involves intensively managed conservation or restoration – but could habitat rewilding based on passive restoration be used to increase biodiversity at lower cost? We analysed the potential of passive lowland agricultural rewilding in England to fulfil the requirements of BNG policy. We considered arable land cover, deer browsing pressure and broadleaved woodland cover as our variables affecting ‘rewilding potential’ and quantified the resulting potential habitat gains using the Biodiversity Metric. We found the likely outcome is mainly habitat restored to poor or moderate condition, and that the southeast part of England has the best rewilding potential, with the eastern side having more potential than the western part of the country. The maximum possible biodiversity units that could hypothetically be generated for different woodland habitat type options varied between 6.0 million and 22.3 million units, in the (albeit highly improbable, and undesirable) case that all arable lowland in England were rewilded. The estimated annual need is currently around 39,000 biodiversity units, which means rewilding a cumulative 0.27–0.90 % of agricultural land back to woodlands starting one year in advance of development could compensate for annual development impacts. A key challenge to this approach is that planners would have to embrace long timescales and uncertainty about the ecological trajectories of habitat offsets.

1. Introduction

The world is facing an environmental crisis, including ongoing climate change, declining biodiversity, and the degradation of

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ecosystem functions and services (IPBES, 2019). Urgent actions are required to prevent further losses, and reverse existing damages. The United Nations has declared the 2020's to be a Decade on Ecosystem Restoration, aiming to halt the loss of biodiversity and prevent climate change. In 2022, the Convention on Biological Diversity agreed a mission to halt and then begin to reverse biodiversity loss by 2030. Countries worldwide have committed to restore a billion hectares of habitats (United Nations Environment Programme, 2021). Many countries are simultaneously implementing net outcomes type biodiversity policies, which require biodiversity impacts to be mitigated, and in some cases over-compensated for to give a net gain in biodiversity (Bull and Strange, 2018), which could therefore potentially support efforts towards nature recovery (Bull et al., 2020).

England is one of the countries with more ambitious targets for habitat restoration and increased woodland cover to combat declining biodiversity and to achieve the 2050 carbon neutrality targets (UK Environment Act, 2021). The aim is to create or restore at least 500,000 ha new wildlife-rich habitat and to increase the woodland cover in England to 16.5 % by the end of 2050 (Defra, 2020). Simultaneously, England has also committed to building 1.5 million new homes over the next five years (UK Government, 2024) which could conflict with the biodiversity and climate targets. The UK Government has launched a mandatory Biodiversity Net Gain (BNG) requirement such that most new development projects in England must increase net biodiversity by 10 % and secure the gains for at least 30 years (Defra, 2020). The change in biodiversity is measured using the habitat-based proxy Defra Statutory Biodiversity Metric ('BM') (Defra, 2024) which quantifies the losses and gains in different habitat types at the project sites in terms of biodiversity 'units'. The biodiversity units are based on the area, habitat distinctiveness and habitat quality with consideration of locality, landscape-scale significance and time to reach target habitat condition (Defra, 2024). The most recent available land use change data in England from 2019 to 2022 indicates that an annual amount of 16,000 ha of previously un-developed land has been converted to new developments (Ministry of Housing, Communities and Local Government (MHCLG), 2023) that could be applicable to the BNG policy meaning a potential need of around 39,000 biodiversity units annually as calculated by the BM in the Discussion section of this paper.

Despite the ambitious biodiversity targets and a mandatory need to create gains, early BNG adoption has so far produced most biodiversity unit gains onsite (improving the habitat quality within or directly-adjacent to the development site). This means decreasing the non-urban habitat area, conflicting with wildlife and ecosystem conservation, and facing a greater risk of being un-enforceable than offsite compensation (zu Ermgassen et al., 2021; Rampling et al., 2023). Producing gains by creating or restoring habitats nearby but offsite could provide a more defensible option for creating biodiversity units that increase the total area of natural habitats with higher distinctiveness while providing more strategically located long-term benefits for biodiversity, despite the available evidence to date pointing to mixed outcomes of offsets (e.g. Maron et al., 2015; zu Ermgassen et al., 2019). However, most biodiversity offsets involve actions that require considerable funds, which ideally would be required for long timescales (as more generally, offset sites should ideally be protected in perpetuity, or at least as long as the impact remains; BBOP, 2012). Hence, offsetting can be expensive and require the organization of long-term management and funding (Damiens et al., 2021).

Rewilding – here understood to mean passive ecological restoration – could potentially be used as a low resource input option for creating offsets if the sites increase their biodiversity value over time by passively restoring themselves. However, key questions in that case would be whether (a) such an approach could generate the necessary habitat gains on the timescales required, and (b) whether policymakers and planners would accept a degree of uncertainty in the ecological trajectory of offset sites. The main idea of rewilding is to restore ecological structures and functions to a level where ecosystems are self-sustaining without continuing human intervention (Perino et al., 2019; Carver et al., 2021). Interacting processes of trophic complexity, dispersal and stochastic disturbance are in the core of resilient and self-organizing ecosystems and should be the main targets of restoration (Perino et al., 2019). Restoring even one of the processes can increase the functionality of others due to their interacting nature meaning the effects of rewilding can also be unexpected.

In this article, we consider ecological rewilding based on natural (i.e. unmanaged) forest regeneration and ecological succession on abandoned agricultural land (see also Pereira and Navarro, 2015). Natural regeneration is the process of forests regrowing from naturally dispersed seeds that germinate and grow on the site (Vieira and Scariot, 2006). Many factors affect the passive restoration and regeneration potential of a certain site (see e.g. Holl and Aide, 2011; Meli et al., 2017; Prach et al., 2020), including soil type and nutrient level (Brudvig et al., 2013), grazing and browsing (Murphy et al., 2022), competitive species (Hudjetz et al., 2014), landscape matrix; presence of target species i.e. seed sources (Cramer and Hobbs, 2007; Broughton et al., 2021; Bauld et al., 2023) and previous land use and its level of intensity (Brudvig et al., 2013; Meli et al., 2017; Reid et al., 2018). Depending on the site characteristics, some restoration activities could be needed to assist the revegetation process, but ecological rewilding aims to minimize human interventions after these initial activities (Navarro and Pereira, 2012) such as planting native tree species or fencing to prevent intensive grazing (Fuentes-Montemayor et al., 2022). Deforested land used for agriculture has been found to naturally regenerate to native woodland globally in many places in only a few decades (Chazdon et al., 2020).

Our aim here is to assess from an entirely hypothetical perspective whether the passive restoration of lowland agricultural land in England would generate substantial biodiversity units within the BNG policy framework – as a preliminary case study illustration of whether passive rewilding might be able to contribute towards biodiversity net outcome policies. We achieve this by calculating the amount of biodiversity units that could theoretically be created through rewilding arable land back to forests based on available seed sources and browsing pressure that allow for the general nationwide analysis. This will provide a rough estimation on the average rewilding potential of English lowland agricultural areas and the potential contribution to BNG policy. Rewilding potentially means less intensively managed – if less certain – offsets in comparison to the highly managed habitat types that can require substantial ongoing restoration and management resources (e.g. creating wetlands). Additionally, it could provide improved biodiversity gains compared to those produced onsite by creating more protected areas and healthy ecosystems that can sustain themselves and their natural processes.

2. Methods

2.1. Study site

We categorised original English habitat types using the map of Terrestrial Ecoregions of the World (Olson et al., 2001), which differentiates 867 ecoregions based on biogeographic units defined by similar species, dynamics and environmental conditions that form distinct assemblages of natural communities. The original broad habitat types that would occur in England would primarily have been lowland beech forests and Celtic broadleaf forests ecoregions, both belonging to the temperate broadleaf and mixed forest biome (Olson et al., 2001). However, agricultural land currently accounts for almost 63 % of the total land area in England (Ministry of Housing, Communities and Local Government (MHCLG), 2020) while forest cover has decreased to only 10 % (UK Environment Act, 2021).

To categorise current English land use, we used the Land Cover Map (LCM) 2015 Version 1.2, 22nd May 2017 produced by the Centre for Ecology and Hydrology (2017). The map included raster layers for 21 different land cover classes that are based on the UK Biodiversity Action Plan Broad Habitat definitions (Jackson, 2000). We chose arable and horticultural land for our analysis, reasoning that this is the most likely land use to be available for passive restoration (unlike e.g. existing urban habitats), and would deliver more gains in biodiversity units compared to existing natural habitats under the BM. Agricultural land can contain valuable biodiversity features e.g. wildflower strips, and arable field margins and hedgerows are considered priority habitats in the UK Biodiversity Action Plan (BAP habitat). We removed these BAP habitats so that they were not calculated as potential restoration areas. Broadleaf woodland which includes the broad habitat of broadleaved, mixed and yew woodland, was chosen for the analysis as the habitat type providing seeds for natural regeneration needed to create the original habitat types based on the terrestrial ecoregions.

Considering the original habitat types present in England (ecoregions of Celtic broadleaf forest and lowland beech forest) we chose the following BM woodland habitat types as the potential habitats resulting from rewilding lowland agricultural areas: *lowland beech and yew woodland* (BAP habitat), *lowland mixed deciduous woodland* (BAP habitat), *other; broadleaved, other; mixed*. Furthermore, the lowland agricultural area was defined as less than 250 m above the sea level, and highlands above that were excluded from the analysis.

2.2. Passive restoration of abandoned agricultural land in England

Ecological rewilding, as conceptualised here, is based on passive restoration via natural regeneration. There is considerable evidence that natural regeneration has occurred on abandoned farmland and resulted in closed-canopy forests in different parts of England in varying time-scales (Harmer et al., 2001; Broughton et al., 2021; Bauld et al., 2023). Abandoned agricultural soil can have native plant species represented in the seed bank, and the land is often suitable for immediate tree growth (Fischer and Fischer, 2012). However, one of the most essential factors affecting woodland regeneration is to have appropriate forest habitat type patches as seed sources near to the target restoration area (Garcia et al., 2010; Bauld et al., 2023). Bauld et al. (2023) found that former agricultural sites in UK were colonized by woody species on a narrow fringe around existing forests and trees. As England's remaining woodlands are fragmented within agricultural landscapes there are multiple seed sources available in adjacent areas: closed-canopy woodland has re-established in 20–50 years on abandoned farmland close to existing woodland in England (Broughton et al., 2021).

2.3. Potential habitat conditions reached through rewilding within the BNG framework

In addition to target habitat type, the BM (Defra, 2024) requires the user to choose the target habitat condition that is measured against the ecological optimum state of the habitat. For woodland types, the condition assessment is based on 13 indicators including browsing pressure, presence of native and invasive plant species, different tree age classes, tree health, amount of deadwood, ground flora, woodland regeneration, vertical structure and disturbance. These indicators are assessed from poor to good by giving 1–3 points accordingly. The total amount of points to reach good condition is set to 33 points whereas moderate condition requires 26 out of max 39 points.

Considering the required criteria for good condition in woodland habitats and the required 30-year timeframe to secure biodiversity gains, it is likely that regeneration through passive restoration reaches poor or moderate condition habitats which is also advised for created woodland sites in the BM user guide (Defra, 2024). In addition, the BM calculation tool includes “time to target condition” that is the average time taken between starting creation or enhancement of habitats and that habitat reaching its target condition. This means a set number of years to reach certain conditions for different habitats. *Other woodland; broadleaved (OW; broadleaved)* and *other woodland; mixed (OW; mixed)* are both set to reach poor condition in 5 years, but moderate condition is reached in 15 years for *OW; broadleaved* and in 30 years for *OW; mixed*. For BAP habitat types poor is reached in 10 years but moderate condition takes already over 30 years. Hence, we tested woodland creation for poor condition BAP habitats, and both poor and moderate condition habitats for the other two woodland types.

2.4. Deer browsing pressure

The adverse effects of dense deer populations on forest vegetation and regeneration are substantial, and well documented in many countries (for review, see Putman et al., 2011). There are six free-living species of deer in England, red deer (*Cervus elaphus*), roe deer (*Capreolus capreolus*), fallow deer (*Dama dama*), sika deer (*Cervus nippon*), Chinese muntjac (*Muntiacus reevesi*) and Chinese water deer

(*Hydropotes inermis*) (Yalden, 1999). Several factors influence browsing pressure in a particular location. These include the species composition of deer, characteristics (biomass, feeding strategy) and population densities of particular deer species as well as the properties of environment. The strongest effect of large herbivores on forest vegetation may be the selective browsing of young tree individuals, which largely determines the composition of mature stand (Skarpe and Hester, 2008). The effects of browsing are very context-specific, which makes it difficult to estimate deer density thresholds beyond which certain desirable woodland regeneration trajectories might become problematic (Table S1). See more details in the Supplementary Materials.

We used the most recent data we could source, the Deer Distribution Survey 2016 data (British Deer Society, 2016), to create a map of deer densities in different parts of England. The survey data were gathered from sightings by the general public and records from major organisations interested in deer. The data were available as jpeg images and we first used them to create presence vs. absence maps. We imported all images to ArcMap (10.5.1) and used the available shapefile of England for digitizing and georeferencing the jpeg images. The deer distribution maps were presented in three categories; data recorded in 2007 and/or in 2011 and reconfirmed in 2016, data unconfirmed in 2016 and data confirmed only in 2016. As the density estimates we used are related to the data from 1995 to 2016, we combined all three categories of deer distribution data in raster maps. Hence, a pixel was assigned a value of 1 if it showed presence of deer in at least one of the three years of records. Next, we reclassified each raster map based on pixel values into binary values (0, 1). This resulted in presence or absence raster maps for each species. Finally, we used the density of each deer species in broadleaved woodland habitats to update the value of presence pixels in deer density maps. For this conversion we used a raster calculator and multiplied the value of presence points by the median of deer density (population density estimates for deer species in broadleaved forests per 100 ha; 5 for red deer, 12 for roe deer, 28 for fallow deer, 7 for sika deer, 23 for Chinese muntjac and 5 for Chinese water deer; Table S1). This resulted in raster maps with the density of deer species as pixel values. As the aim of this analysis was the deer density (all species combined) in each pixel, we combined all maps to make a sum deer density map, applying the data related to all species (see Fig. S1).

2.5. Rewilding potential

We selected three main variables for scoring of the sites: arable land cover, forest cover (i.e. forest proximity) and deer browsing pressure. Again, to reiterate the rationale of this hypothetical rewilding analysis: we chose arable land due to the great prevalence of agricultural land with low biodiversity values in England. Forest cover is necessary for passive restoration as natural regeneration requires dispersed seed that are produced by nearby forests. Deer browsing pressure is justified due to its well-documented impact on forest vegetation (e.g. Putman et al., 2011).

We developed an index to act as a proxy for the rewilding potential of different pixels, based on the three central variables in our analysis (current land cover type, proximity to woodland fragments, combined deer density). Rewilding potential means how well the passive restoration is likely to succeed with minimal initial actions such as seeding, planting or fencing from browsing. Table 1 presents the different variables and categories for scores from 1 to 3; higher scores indicate greater rewilding potential. The forest cover ranged between 0 % and 38 % in a full-size (100 km²) pixel and we scored the categories by dividing the highest cover score (38 %) by the number of categories (n = 3). Hence, areas with the best rewilding potential have over 25 % forest coverage, the least potential have less than 13 %, while the moderate potentiality is between. The remaining forests in England are widely distributed and fragmented inside most pixels. Therefore, high forest cover inside a pixel means there are more likely seed sources close to arable land which improves the restoration potential. Low forest cover on the other hand reduces the overall regeneration potential, but it also lowers the chance of woodlands being close to arable land.

Arable land cover varied in a pixel from 0 % to 95 %, and the scoring was decided as for forest cover (above). Pixels with least potential had smaller than 32 % arable land cover due to smallest possible restoration area whereas pixels with greater than 63 % arable land cover were categorised as most potential areas. This was based on the largest possible restoration area in a pixel. Despite high arable land cover, small and fragmented forest patches can still provide seed sources in these pixels.

The deer browsing pressure scoring is based on the information provided in Table S1. The best areas for rewilding would have minimal browsing pressure and therefore areas with a population density of less than 5 gives the highest scores. Areas with least potential had deer population density of over 25 for that being the lowest certain threshold density for smaller deer species (roe deer < 25, muntjac deer < 50).

After categorizing the habitats, we used QGIS 3.22 to calculate the combined scores of the deer browsing pressure, forest and arable land cover for each pixel. We categorised the resulting scores of 3–9 again in the following categories: 3–4 = poor, 5–7 = moderate and 8–9 = good rewilding potential. Good rewilding potential areas are considered to revegetate without initial actions, areas with moderate potential are likely to need some initial actions and areas with poor potential require an establishment of additional seed

Table 1

Scoring of the sites based on deer browsing pressure and the amount of forest and arable land cover in a 10 km × 10 km pixel. The variables are given scores from 1 to 3 based on the rewilding potential.

Deer browsing pressure		Forest cover		Arable land cover	
Threshold population density per 100 ha	Score	Cover (%)	Score	Cover (%)	Score
> 25	1	< 13	1	< 32	1
5–25	2	13–25	2	32–63	2
0–5	3	> 25	3	> 63	3

sources and/or protection from browsing. We calculated the amount of units created for habitats reaching poor and moderate condition applying the area sizes of rewilding potential based on our analysis.

We also performed a sensitivity analysis upon our index, testing the uncertainty in resultant forest vs arable land cover achieved by changing the percentages needed to obtain different scores (Table S2). We increased and decreased each of the percentage categories individually by 5 % for forest and 10 % for arable land cover because of their different maximum coverages (38 % forest, 95 % arable land). We cross tested all different scenarios, creating 8 different versions (V1–V8).

2.6. Calculation of the biodiversity units for BNG

The BM is implemented in a habitat-based calculation tool that scores different habitat types and the resulting biodiversity units from the habitat degradation, enhancement or creation. The BM calculates the relative value of site’s biodiversity as biodiversity units that are based on the change of habitat area and quality, which is measured by the habitat distinctiveness, condition and strategic significance. Distinctiveness is based on the habitat type and its distinguishing features such as species richness and rarity. Scores for distinctiveness are pre-determined for different habitat types in 6 values ranging from 0 and 1 (very low), 2 (low), 4 (medium), 6 (high) to 8 (very high). Table S3 summarizes all input categories and scores. The condition score is based on key physical features and habitat’s ability to support typical species and hence chosen by the BM user from 1 (poor), 2 (moderate) and 3 (good) with also an option of N/A and 0. Strategic significance is about habitat’s local significance depending on the habitat type and location. It is based on the Local Nature Recovery Strategy (LNRS) and chosen from 3 options (1–1.15). In case of unpublished LNRS, alternative documents should be used but medium strategic significance can be assigned when the user has demonstrated the habitat’s significance in that certain location. Furthermore, the BM incorporates three risk multipliers associated with habitat creation and enhancement. Difficulty of creation or enhancement multiplier for different habitat types is a fixed value. However, the temporal risk can be reduced when habitat creation or enhancement is started in advance of the development. Spatial risk categories on the other hand are considered for off-site habitat creation and enhancement in 3 levels depending on the distance between offset and development. However, only developer will apply the spatial risk multiplier (SRM) when the location of the development is known while for the producer it is a fixed value of 1 meaning local offset. A key principle in the BNG policy is that compensation is local to the impacted site to ensure that offsets are likely to be as similar as possible to the lost habitats. Hence, less units are produced by using SRM of 0.75 or 0.5 when offsetting is implemented outside the local planning authority (LPA) or national character area (NCA). The Excel-based tool requires the user to choose the habitat type, area, condition, strategic significance, spatial risk category and if the habitat is enhanced/created in advance or it is delayed.

In the tool, we used the D-1 Off-Site Habitat Baseline and D-2 Off-Site Habitat Creation to calculate different options of the biodiversity units produced by rewilding arable land to forest. We chose cropland as the D-1 Off-Site Habitat Baseline and for D-2 we used different woodland types (see Section 2.1) as the proposed habitat to be created offsite. We tested how many biodiversity units would be created if baseline habitat was rewilded to poor or moderate condition woodland habitat. For strategic significance we chose

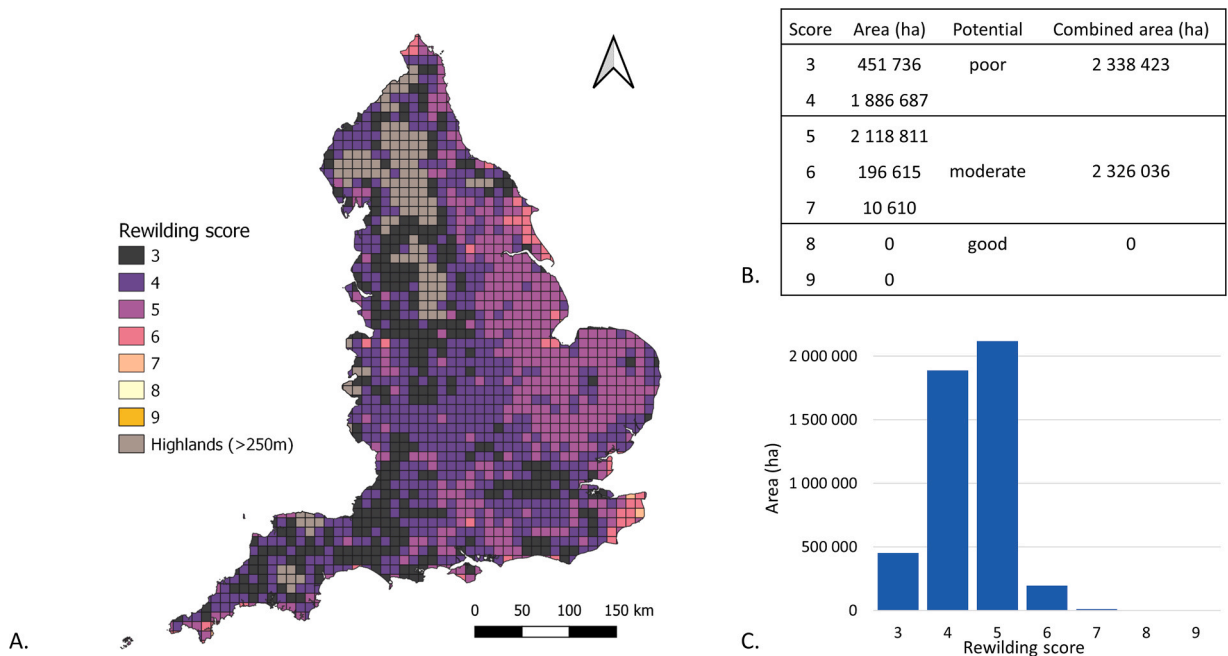


Fig. 1. A.) Map of England showing the rewilding potential of lowland agricultural land as scores across the country. B.) Table with area (ha) of different rewilding potential scores from 3 (poor potential) to 9 (good potential). C.) Figure of rewilding scores and associated areas (ha).

the average option, meaning the location is ecologically desirable but not identified in local plans. We considered choosing medium option would likely produce average results as we were not able to study the actual local strategies of different places across England. We tested habitat creation in advance for 1, 5 and 10 years before impacts on the habitat it is compensating for as restoration activities for offsets are encouraged to be started before impacts to ensure required gains are achieved (Maron et al., 2012).

3. Results

3.1. Rewilding potential

Based on our criteria, there are no areas belonging to the maximum rewilding potential category (scores 8–9). Instead, we found the highest score to be 7 (10,610 ha), which is part of the moderate potential category (scores 5–7) (Fig. 1.). There is an almost equal amount of areas with poor (2,338,423 ha) and moderate (2,326,036 ha) potential, but the most frequent score is moderate category 5 (2,118,811 ha).

Fig. 1 presents the spatial distribution of our rewilding index score across England. According to our criteria, rewilding potential is concentrated to the eastern part of England with the best (scores 6–7) potential areas located on the northeast and southeast coast (mainly in Kent) with a few separate cells in the west. Areas with least rewilding potential (scores 3–4) are located mainly in the southwest England (excluding the south coast), western England around the Pennines highlands and the London area.

3.2. Biodiversity units

We calculated the generated biodiversity units based on our rewilding potential analysis. Hence, we calculated options for habitats reaching poor and moderate condition and the related areas (ha) for potential habitat types after passive restoration with different options for habitat creation in advance (years). We assumed that poor potential areas would more likely achieve poor target condition and moderate potential moderate condition. However, reaching these conditions is partly based on the amount of initial management actions to reach the target. The results for different scenarios are shown in Table 2. Both BAP habitats are high distinctiveness habitats, creating the same amount of biodiversity units and therefore presented in the same column.

Table 2 shows the maximum amount of biodiversity units – if all arable land (4,664,459 ha) in England were rewilded – is 22.3 million, if 50 % of the area is restored to moderate condition *OW; broadleaved* in local area 10 years in advance and 50 % to poor condition BAP habitat with the same options. The minimum positive amount is 6.0 million units, if 50 % arable land is restored to moderate condition *OW; mixed* habitat of poor condition starting one year in advance and 50 % to poor habitat condition in either *OW; mixed* or *OW; broadleaved* with the same options. However, two options for BAP habitats (created 1 and 5 years before development) result in negative numbers, meaning these habitat types with this target condition cannot be used in the system unless they have been created already 10 years before the development project starts. Overall, the highest amount is generated when habitat restoration starts 10 years in advance, and the minimum amount of units is produced when restoration starts one year in advance. The positive biodiversity units produced per ha vary from 0.93 to 5.16 with an average of 2.6 units/ha. The most productive habitat type per ha is *OW; broadleaved* being restored to moderate condition (max. 5.16 units/ha) while the least productive is *OW; mixed* in moderate condition (max. 2.12 units/ha). However, the max units could be higher if restoration started earlier whereas *OW; broadleaved* and *OW; mixed* reaching poor condition results in only 2.2 units/ha. Additionally, restoration to poor condition BAP habitat generates the least amount of units in the first years, but when reached the poor condition (habitat created 10 years in advance) it outperforms the units produced by the poor condition of the other two habitat types. All generated biodiversity units/ha (habitat unit net change/ha) for offsets created in advance for years 1–10 are shown in Fig. S2.

Table 2

Generated offsite biodiversity units for all lowland arable land in England calculated by biodiversity metric using cropland as baseline habitat. The table shows biodiversity units for different woodland habitat type areas that are restored to poor or moderate condition and the final time to target condition.

Proposed habitat	Lowland beech and yew woodland, Lowland mixed deciduous woodland (BAP habitat)			Other woodland; broadleaved			Other woodland; mixed								
	Poor	Moderate	Poor	Moderate	Poor	Moderate	Poor	Moderate	Poor	Moderate	Poor	Moderate	Poor		
Target condition	Poor			Moderate			Poor			Moderate			Poor		
Area (ha)	4,664,459			2,326,036			2,338,423			2,326,036			2,338,423		
Habitat created in advance (years)	1	5	10	1	5	10	1	5	10	1	5	10	1	5	10
Final time to target condition (years)	9	5	0	14	10	5	4	0	0	29	25	20	4	0	0
Habitat unit net change (million)	-2.9	-1.8	20.5	7.3	9.2	12.0	3.8	5.1	5.1	2.2	3.3	4.9	3.8	5.1	5.1
Habitat unit net change/ha	-0.62	-0.38	4.4	3.14	3.96	5.16	1.62	2.2	2.2	0.93	1.41	2.12	1.62	2.2	2.2

3.3. Sensitivity analysis

The sensitivity analysis produced 8 different scenarios shown in Table 3 with comparison to the original result (V0). The original result (V0) has the most equal scoring between poor and moderate area, and the result falls in the middle of the different versions tested in the analysis. Only V5 (Table 3) includes areas with good rewilding potential. We also tested the spatial distribution of the rewilding potential and found the potential is concentrated to eastern England in all scenarios (Fig. S3). See full details in Supplementary materials.

4. Discussion

In this paper, we have modelled the potential of rewilding (understood as passive habitat restoration) applied in lowland agricultural areas of England, in relation to an explicit biodiversity policy target (BNG), and on existing ecological knowledge around cost-efficient habitat restoration methods (Myers and Harms, 2009).

We can compare the amount of generated units with the estimated need from development projects. The English Ministry of Housing, Communities and Local Government (2023) estimates that 48,500 ha of agricultural and other undeveloped land in England has been built-on between 3-year period of 2019–2022 which means at least 16,167 ha of habitats would need to be compensated for per year. Making assumptions about the associated pace of land use change, we can use the BM to calculate the potentially necessary biodiversity units. Assuming agricultural land (habitat of low distinctiveness score 2, fixed condition score 1, medium strategic significance) is the most often changed land-use type for development (developed land; sealed surface with very low distinctiveness score 0), 35,567 biodiversity units would be lost annually, and when adding the 10 % biodiversity gain required by the BNG policy, the overall amount of required units is 39,124 annually. In our analysis, 1 ha of arable land restored to *OW*; *mixed* in moderate condition generates the lowest positive amount of units (1 ha = 0.93 biodiversity units) when restoration is started one year in advance of the impact (Table 2). Therefore, 42,069 ha of arable land would need to be rewilded annually to offset for the development projects. This means a cumulative 0.90 % of all lowland agricultural land in England would need rewilding to offset for annual development projects. However, if restoration were started a year in advance but the target habitat type is *OW*; *broadleaved* in moderate condition that generates 3.14 biodiversity units/ha, only 12,460 ha, cumulative 0.27 % of arable land would need rewilding. The largest amount of units created is 5.16/ha which means rewilding a cumulative 0.16 % when restoration is started 10 years in advance. However, in practice it is more likely that restoration is started only a year in advance when the need for rewilding would be 0.27–0.90 % per year, and for poor target condition *other woodland* habitat types 0.52 %. However, if some of the developed area is of moderate or high distinctiveness habitats, then the need for biodiversity units is also higher. Nonetheless, most projects proposals of the councils that were early adopters of BNG were found to clear low distinctiveness habitats such as cropland and modified grassland (zu Ermgassen et al., 2021; Rampling et al., 2023). Hence, offsetting could be done via the woodland habitat types used in our analysis. Another important factor to consider is the SRM of 0.5 or 0.75 that increases the amount of required biodiversity units for offsets that are outside the same LPA or NCA as development. Therefore, if some of the offsets are not local the needed area for rewilding could eventually be higher than the options calculated in this analysis. Nevertheless, based on our results and the estimated need for biodiversity units, rewilding of the type envisioned here could generate a considerable amount of biodiversity units with some initial actions to advance the rewilding process and the right target condition for habitat types especially with local offsets created in advance of the development.

Although rewilding could generate a significant amount of biodiversity units, it could be problematic to estimate the target habitat type correctly. In addition to target habitat type, the BM requires its user to choose a target habitat condition which we tested for the most likely outcomes in the given timeframe. However, it is possible to aim for good condition habitats with a legal agreement that secures habitats for more than the minimum 30-year project timeframe (Defra, 2024). Accordingly, monitoring the short- and long-term impacts of rewilding is crucial for the BNG results, and the establishment of national offset registers with monitoring information are also required to confirm biodiversity targets have been achieved (Kujala et al., 2022). Furthermore, Rampling et al.

Table 3

Sensitivity results V1–V8 and the original result V0. F-/ + means forest cover % change and A-/ + arable land cover % change tested for sensitivity.

	Version 0 (ha)	V1 (ha)	V2 (ha)	V3 (ha)	V4 (ha)	V5 (ha)	V6 (ha)	V7 (ha)	V8 (ha)
%-change	Original	Forest (F) – 5 %	F + 5 %	Arable (A) – 10 %	A + 10 %	F – 5 %, A – 10 %	F – 5 %, A + 10 %	F + 5 %, A – 10 %	F + 5 %, A + 10 %
Score									
3	451,736	255,624	544,781	207,778	817,633	120,861	493,473	248,667	973,660
4	1,886,687	1,512,914	1,996,248	1,327,386	2,351,226	979,741	2,056,207	1,487,875	2,347,426
5	2,118,811	2,468,865	1,933,899	2,807,782	1,337,068	2,778,364	1,898,842	2,676,721	1,189,931
6	196,615	395,311	179,473	294,804	157,893	743,496	209,078	225,796	153,354
7	10,610	31,746	10,059	26,710	639	35,777	6,859	25,400	88
8	0	0	0	0	0	6,221	0	0	0
9	0	0	0	0	0	0	0	0	0
Potential									
Poor	2,338,423	1,768,538	2,541,029	1,535,164	3,168,859	1,100,602	2,549,680	1,736,542	3,321,086
Moderate	2,326,036	2,895,922	2,123,431	3,129,296	1,495,600	3,557,637	2,114,779	2,927,917	1,343,373
Good	0	0	0	0	0	6,221	0	0	0

(2023) suggest establishing habitat banks to create biodiversity gains in advance in England.

We tested the amount of units produced for habitat restored to moderate and poor condition which could be expected for the analysed woodland types during the 30-year timeframe. The development of good condition woodland habitats as described in the BM user guide on the other hand could take decades to evolve. Fuentes-Montemayor et al. (2022) studied old tree planting sites on former agricultural land in UK and found they can take 80–160 years to develop certain similar vegetation attributes present on semi-natural ancient woodland sites. They suggest that promoting natural regeneration and reducing browsing pressure could accelerate this transition to a more biodiverse forest structure as present in older woodlands. Naaf and Kolk (2015) have also found over a century long time scales for post-agricultural temperate forest sites to be recolonized by forest specialist species. Bauld et al. (2023) found that in former agricultural land in England a tree density of 100/ha occurred approximately after 19 years, and natural woodland species colonized the area only 135 m from seed sources. Even though complete canopy cover could be established in 20–30 years, many local woodland plants can take over a century to establish populations in the regenerated forest (Harmer et al., 2001). A case study example by Natural England (2024) illustrates how rewilding projects could achieve good condition and high distinctiveness habitats (BAP habitats) by applying a second phase i.e. calculating another 30-year period after the first targets have been achieved.

The rewilding potential in our study is based on a large-scale nationwide analysis that can provide rough estimates on the average rewilding potential of the lowland agricultural areas. Because of this wide scale and large areas of agricultural land that are assumed to be revegetated, there are less likely enough seed sources nearby to provide good rewilding potential. However, there are likely agricultural sites with good rewilding potential when studied on a smaller agricultural site level and when less arable land cover is required (Table 3). The agricultural sites estimated with poor and moderate rewilding potential in our analysis would need some planting or seeding in the beginning to reach the target condition and develop into a closed-canopy woodland. Rey Benayas and Bullock (2012) suggest planting dense forest patches of native tree species to boost the passive restoration of vast agricultural areas. Rewilding commonly involves active management in the short-term (Thierry and Rogers, 2020) and active and passive approaches are recommended to be used together to meet the targets with reasonable financial costs and time scales (Prach et al., 2020; Bauld et al., 2023). Furthermore, Prach et al. (2020) argue that the most successful passive restoration sites are small, mildly disturbed, have moderate productivity, and are under low abiotic stress in a moderately natural landscape. In case of woodlands, the rewilded area should also be large enough to enable some permanent open space areas (Harmer et al., 2001). Nevertheless, there is evidence that passive restoration is a low-cost option in expanding native woodlands in relatively short timescales with associated ecosystem services providing significant returns for low levels of investment (Myers and Harms, 2009; Navarro and Pereira, 2012; Broughton et al., 2021). However, the timescales can vary as in a study by Harmer et al. (2001) where woody species colonized two abandoned farmland sites in 10 and 30 years. As Rampling et al. (2023) suggest moving to offsets could improve ecological outcomes of BNG policy but require increased investments to conservation actions on private land, rewilding could provide a reasonable solution even when some initial actions were required, and this opportunity should be further investigated in detail.

Our study is a simplified model of the rewilding potential, being limited by unknown soil type and nutrient status which have been, among precursor vegetation, suggested as factors affecting the regeneration potential needed for rewilding (Harmer et al., 2001). However, we have modelled different woodland habitat types in the BNG calculations as the nutrient requirements vary between habitats. Also, soil type is more likely to affect the time required to a closed-canopy forest instead of the site not forming forest structures (Harmer et al., 2001). We did not test climatic variation which could affect the woodland habitat succession on the site, but the climate conditions in England are rather constant throughout the country (Panks et al., 2022). The deer density in our analysis could be overestimated as the data we used were given in 10 km squares to keep real deer locations protected (British Deer Society, 2016). However, 40 % of woodland habitat in UK is in unfavourable condition because of herbivore damage (Forestry Commission, 2020).

The next step in rewilding would be to study different land use options and landowners' preferences as an important aspect of rewilding is the consideration of needs and expectations of different stakeholders (Perino et al., 2019; Carver et al., 2021) and the overall willingness of landowners to participate in rewilding. It would also be essential to analyse the more realistic rewilding potential on a smaller scale because of the various local factors affecting it. This would include especially the soil quality of agricultural lands close to forests and other seed sources. Spatial analysis of rewilding should also include a connectivity analysis to existing woodlands and conservation areas to create habitat networks. Considering the rewilding potential of agricultural sites close to human settlements would be important as offsets are often established further from people than the development sites especially in urban areas despite the multiple benefits that nearby nature creates to society (Kalliolevo et al., 2021), and rewilding for BNG should consider the availability of greenspace and the increase of recreational areas for people (Natural England, 2021). A strategic approach to targeting offsets could provide improved benefits for biodiversity and ecosystem service delivery (Mancini et al., 2024), although at the same time would move away from the proximity principle long established as part of good biodiversity offset practice.

Despite the UK target of a 10 % net gain in biodiversity for all new infrastructure projects, it is unclear how well the policy will work in practice (zu Ermgassen et al., 2022). Crucially, it is always preferable to avoid in the first place clearing habitats and especially those habitats that are still in their natural state (Bull et al., 2022). New development projects should be directed to low-quality and already damaged habitats or to built-up areas while funding for rewilding and other restoration measures should be established and supported. But our work here explores quantitatively for the first time whether a programme of rewilding through passive restoration might be one possible input to realise the necessary gains under a no net loss/net gain biodiversity policy, and we find – at least for the UK case study – there is considerable potential to do so.

Ethics Statement

Not applicable: This manuscript does not include human or animal research.

Declaration of Competing Interest

Authors have nothing to declare.

Acknowledgements

We thank Chris Sandom and Matti Salo for their comments, and Roshan Sharma, Sophus zu Ermgassen and Erica Marshall for technical support. We also thank two anonymous reviewers for their valuable comments. HK was funded by the Kone Foundation and The Doctoral Programme in Biology, Geography and Geology in University of Turku.

Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at [doi:10.1016/j.gecco.2025.e03603](https://doi.org/10.1016/j.gecco.2025.e03603).

Data availability

Data will be made available on request.

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