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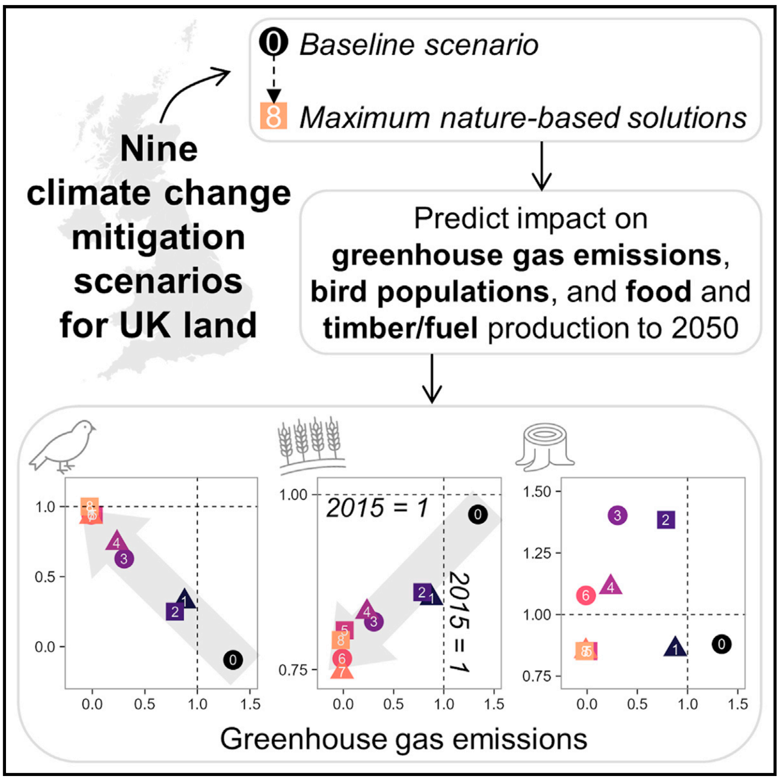
Spatially targeted nature-based solutions can mitigate climate change and nature loss but require a systems approach

Graphical abstract



Highlights

- We compared nine exploratory scenarios of future UK land use
- Scenarios with more nature-based solutions minimized greenhouse gas emissions
- Habitat for breeding birds is predicted to increase under these "net zero" futures
- Trade-offs with food production can be managed via supply- or demand-side changes

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In brief

Halting climate change requires net greenhouse gas (GHG) emissions to at least fall to zero. UK land (including agriculture, forestry, and peatlands) is currently a net emitter of GHGs. Reducing these emissions entails trade-offs. By modeling nine future scenarios of UK land use that deployed different amounts and combinations of measures aimed at reducing emissions and increasing carbon drawdown, we find that a net zero UK land sector is possible. However, this requires strategic land-use planning and food system changes.

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Spatially targeted nature-based solutions can mitigate climate change and nature loss but require a systems approach

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SCIENCE FOR SOCIETY Halting climate change requires net greenhouse gas (GHG) emissions to at least fall to zero. Healthy ecosystems can capture and store carbon, meaning land management can contribute to "net zero" efforts. However, the UK land sector—including agriculture, forestry, and peatlands—is currently a net GHG emitter. Reducing land emissions requires changing the way we use land, such as by increasing tree cover, restoring peatlands, and using agricultural inputs more sparingly. These changes have impacts beyond GHGs. For example, the creation and restoration of natural habitats should benefit biodiversity, but at the cost of some food production. This can be compensated for by reducing food waste, improving productivity, and reducing meat and dairy consumption, but a strategic approach to future land use is needed to balance these outcomes. Business as usual is not an option, given the UK land sector's ongoing contribution to climate change, which itself presents a risk to farming.

SUMMARY

Finite land is under pressure to provide food, timber, human infrastructure, climate change mitigation, and wildlife habitat. Given the inherent trade-offs associated with land-use choices, there is a need to assess how alternative land-use trajectories will impact the delivery of these benefits. Here, we develop nine exploratory, climate change mitigation-driven land-use scenarios for the UK. The scenario that maximized deployment of nature-based solutions reduced greenhouse gas (CO_2e) emissions from the land sector by >100% by 2050 but resulted in a 21% decline in food production. All mitigation scenarios delivered aggregate increases in habitat availability for 109 bird species (including 61 species of conservation concern), although farmland-associated species lost habitat. Our study reiterates the potential of nature-based solutions to address global climate and biodiversity challenges but also highlights risks to farmland wildlife and the importance of food system reform to mitigate potential reductions in primary food production.

INTRODUCTION

Land use and food systems have key roles to play in addressing global challenges, and land is under pressure to deliver (among other things) food, timber and fuel production, human infrastructure, climate change mitigation, and biodiversity conservation.[1–3] Some land-based actions can simultaneously meet multiple challenges. For instance, the effective protection and restoration of natural ecosystems can support biodiversity conservation and carbon storage and sequestration,[4–6] as well as a range of other beneficial ecosystem services.[7,8] Despite these opportunities for synergies, some trade-offs are unavoidable; climate change mitigation actions sometimes conflict with biodiversity conservation aims,[9] and vice versa[10]; carbon storage and wildlife abundance are often negatively associated with agricultural yields[11,12]; and spatial congruence between priority areas for carbon sequestration and biodiversity conservation is often lacking.[1,13,14]

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Navigating these trade-offs across finite land area is a critical societal challenge.

Under the 2015 Paris Agreement of the United Nations Framework Convention on Climate Change, 196 nations agreed to set ambitious nationally determined contributions toward keeping "global average temperature to well below 2°C above pre-industrial levels and to pursue efforts to limit the temperature increase to 1.5°C above pre-industrial levels."[15] The parallel Convention on Biological Diversity has agreed the Kunming-Montreal Global Biodiversity Framework, which aims to halt and reverse the loss of biodiversity by 2030 and includes a target to restore at least 30% of degraded ecosystems by 2030 (Target 2) and to minimize the impact of climate change and climate action on biodiversity (Target 8). While global studies have estimated the potential of measures such as ecosystem protection and restoration to address these grand challenges,[1,4,16] national studies can identify finer-scale constraints, and thus provide a more realistic estimate of the size of the opportunity presented by ecosystem restoration.[17,18]

Here, we focus on the UK, an example of a country whose government has committed to achieving "net zero" GHG emissions by 2050, meaning that any residual emissions (measured in CO_2-equivalents [CO_2e] according to global warming potential over a 100-year time frame [GWP_{100}]) should be balanced by carbon removal in the same year. In their most recent sixth carbon budget, the UK's Committee on Climate Change (CCC) presented cross-sector scenarios for reaching net zero.[19] Their most ambitious scenario (*Tailwinds*) sees the land sector (that is, Agriculture, Forestry, and Other Land Use; also referred to as Agriculture and Land Use, Land-Use Change, and Forestry) reach net zero by 2046, with net sequestration of 14 Mt CO_2e by 2050, all while maintaining per capita food supply (albeit predicated on ambitious increases in crop and livestock yields).

Nature-based solutions (NBSs) present an opportunity for these climate change mitigation pathways to deliver increases in the quality and quantity of natural and semi-natural habitats, thus contributing toward the UK government's commitments on nature recovery.[20] There are also risks. Poorly planned woodland creation can result in reductions in functional habitat area for species of non-woodland habitats,[21] and even well-planned woodland creation results in species losses as well as gains.[22] The CCC pathways are also heavily reliant on crop yield growth (34%–59% by 2050), which, alongside food waste reduction and dietary change, are required to spare substantial areas of land from agricultural use. While this land-sparing approach is likely to benefit many bird species, it would have a negative impact on birds associated with lowland agriculture,[23] a group of conservation concern in the UK.[24] In addition, stated emissions reductions would be diminished if such ambitious levels of yield growth were to prove unachievable, or if the surplus land remained under agriculture.

Given the inherent trade-offs associated with land-use choices, there is a need to assess how alternative future land-use trajectories will impact the delivery of the services and products derived from UK land. Previous efforts have tended to focus on only one or two scenarios, interventions, or outcomes, or have lacked an explicit spatial element, precluding the consideration of some outcomes that are inherently spatial (e.g., Lamb and co-workers,[23,25] Jungandreas et al.,[26] Smith et al.,[23,25–28] and Redhead et al.[23,25–28]). We see a particular need to (1) explore scenarios that involve greater emphasis on NBSs compared with the CCC[19] and (2) assess the likely biodiversity impacts (winners and losers) of climate change mitigation within the UK land sector.

We use scenario modeling to evaluate the trade-offs and synergies among climate change mitigation, nature conservation, and food production under nine alternative land-use futures for the UK. Our approach is spatially explicit (25-m resolution) to account for the finite and heterogeneous nature of land. By focusing on a single country, we adopt a much finer resolution of spatial analysis than is feasible for global studies (e.g., Strassburg et al.[1]) and consider issues that are of particular importance in a national context (e.g., protecting internationally important populations of breeding wading birds). Finally, rather than indirectly inferring or extrapolating biodiversity impacts (e.g., Powell et al.[29,30] and Smith et al.[29,30]), we use an ensemble approach to predict changes in habitat availability for individual bird species, allowing species-specific responses of different scenarios to be decomposed. We show that, through ambitious deployment of NBSs alongside other mitigation measures, a net zero UK land sector can be achieved while delivering an aggregate increase in habitat availability for breeding birds.

RESULTS

Methods summary

Our land-use scenarios are designed to be indicative rather than prescriptive or predictive, and include both on-farm and off-farm interventions, because both are likely to be important for biodiversity conservation.[31,32] We consider 10 climate change mitigation measures (Table 1). Mitigation measures involve modification of land cover (e.g., conversion to woodland, semi-natural grassland, semi-natural wood pasture, fen, bog, or dedicated biomass crops), the addition of trees and shrubs to existing farmland (through silvopastoral or silvoarable agroforestry and hedge creation), improved crop or livestock management through low-carbon farming practices and organic farming, and improved condition of degraded peatlands. Mitigation measures are deployed between 2020 and 2050, after which land cover and management are held constant. For each measure we define unsuitable land types (e.g., no woodland creation on existing priority habitats, designated sites, peat soils, or carbon-rich organomineral soils[33]), generally avoiding competition with the most productive farmland and promoting habitat creation near to existing patches of the same habitat. We primarily report results for the year 2050 (the UK net zero policy target), but project GHG emissions and timber and biomass fuel production to 2100.

To obtain a manageable number of illustrative scenarios, we consider two to four discrete ambition levels for each measure (Table 1). This still leaves >45,000 potential combinations of each measure and ambition level. Rather than attempt to exhaustively sample from within this option space, we instead develop nine exploratory scenarios that reflect alternative potential pathways for future land use in the UK (Figure 1). We first consider a *Baseline* scenario, which reflects 2015 land cover (Land Cover Map 2015[35]) and involves no additional mitigation.

Table 1. Summary of mitigation measures and ambition levels

Mitigation measure	*Medium*	*High*	*High+*
Intertidal habitat re-creation[a]	–	0.48 kha/year	–
Peatland restoration[b]	all upland peat restored by 2045; all extraction halted by 2035; 10% of lowland cropland to paludiculture and 30% under raised water tables by 2050; 20% of forestry on peat removed by 2035	*Medium* + 50% of lowland grassland restored by 2050; 25% of lowland cropland restored, 15% to paludiculture and 35% under raised water tables by 2050	*Medium* + 75% of lowland grassland restored and 25% to paludiculture by 2040; 50% of lowland cropland restored and 50% to paludiculture by 2040; 100% of forestry on peat removed by 2050
Woodland creation[c]	30 kha/year by 2025, rising to 50 kha/year 2035–2050; 66% broadleaved	30 ka/h by 2025, rising to 50 kha/year by 2030, then 70 kha/year 2035–2050; 66% broadleaved	–
Agroforestry[d]	11 kha/year silvoarable, 16.7 kha/year silvopasture	11 kha/year silvoarable, 25 kha/year silvopasture	–
Wood pasture[e]	16.7 kha/year on current grassland	25 kha/year on current grassland	–
Semi-natural grassland creation[f]	6 kha/year on current grassland/cropland	15 kha/year on current grassland/cropland	–
Hedge creation[g]	40% increase in hedge length by 2050	50% increase in hedge length by 2050	–
Biomass crops[h]	10 kha/year of *Miscanthus* by 2031	*Medium* + 10 kha/year each of SRC and SRF by 2031	–
Organic farming[i]	–	25% of farmland converted to organic by 2035	–
Low-carbon farming[j]	behavioral and innovative measures, improved nitrogen use efficiency (10% for grass, 20% for crops), electrification of machinery and power	*Medium* + higher uptake of behavioral measures; crop nitrogen use efficiency improvement increases to 30%	*High* + additional sustainable practices

Low ambition (not shown) involves no change in land cover or land management from the baseline.
[a]Not considered by the CCC.[19]
[b]*Medium* ambition reflects CCC.[19]*Headwinds*, but without conversion of lowland grassland or cropland to wetland habitat. *High* ambition reflects CCC.[19]
Balanced Net Zero Pathway/Widespread Engagement. *High+* ambition reflects full rewetting of lowland peat (to either wetland or paludiculture), and full restoration of all forestry on peat.
[c]*Medium* ambition reflects CCC.[19]
Balanced Net Zero Pathway. *High* ambition reflects CCC.[19]
Widespread Engagement.
[d]*Medium* ambition reflects CCC.[19]
Balanced Net Zero Pathway. *High* ambition reflects CCC.[19]
Widespread Engagement.
[e]Not considered by the CCC.[19]
[f]Not considered by the CCC.[19]
[g]*Medium* ambition reflects the CCC.[19]
Balanced Net Zero Pathway/Widespread Engagement. *High* ambition delivers additional hedge creation.
[h]*Medium* ambition reflects CCC.[19]
Widespread Engagement. *High* ambition reflects CCC.[19]
Balanced Net Zero Pathway. SRC, short-rotation coppice; SRF, short-rotation forestry.
[i]Not considered by the CCC.[19] *High* ambition reflects EU Farm to Fork Strategy: https://food.ec.europa.eu/horizontal-topics/farm-fork-strategy_en.[34]
[j]*Medium* ambition reflects CCC.[19]
Balanced Net Zero Pathway. *High* ambition reflects CCC.[19]
Widespread Engagement. Behavioral and innovative measures are as defined by the CCC,[19]
and include cover crops, grass/legume mixes, grass leys, precision livestock feeding, enhanced livestock breeding, and feed additives. *High+* introduces additional sustainable practices (loosening compacted soils and more legumes in crop rotations).

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Scenario name	Scenario description	Intertidal habitat creation	Peatland restoration	Woodland creation	Agroforestry	Wood pasture	Semi-natural grassland creation	Hedge creation	Biomass crops	Organic farming	Low carbon farming
0: Baseline	No change in land cover or land management	Low	Low	Low	Low	Low	Low	Low	Low	Low	Low
1: On Farm Measures (Organic)	Scenario 7, without 'off-farm' measures	Low	Med.	Low	High	Low	Low	High	Low	High	High+
2: On Farm Measures (Balanced)	Scenario 3, without 'off-farm' measures	Low	Med.	Low	Med.	Low	Low	Med.	High	Low	Med.
3: Balanced Pathway	Based on CCC '*Balance Net Zero Pathway*'	Low	High	Med.	Med.	Low	Low	Med.	High	Low	Med.
4: Widespread Engagement	Based on CCC '*Widespread Engagement*'	Low	High	High	High	Low	Low	Med.	Med.	Low	High
5: NBS	Scenario 6 without biomass crops	High	High+	High	High	Med.	Med.	High	Low	Low	High+
6: NBS with biomass crops	Scenario 4 + additional nature-based solutions	High	High+	High	High	Med.	Med.	High	Med.	Low	High+
7: NBS with organic farming	Scenario 6 + organic farming	High	High+	High	High	Med.	Med.	High	Low	High	High+
8: NBS extra	Scenario 6 + additional wood pasture and semi-natural grassland	High	High+	High	High	High	High	High	Low	Low	High+

Figure 1. Summary of scenarios
Green-shading indicates the level of ambition for each mitigation measure (columns) under each scenario (rows). NBS, nature-based solutions; CCC, Committee on Climate Change.

We then reproduce two CCC pathways (*Balanced Pathway* and *Widespread Engagement*), which we use as a starting point from which to increase or decrease ambition for individual measures. Neither of these CCC pathways result in a net zero land sector by 2050,[19] so *NBS with biomass crops* sees increased deployment of NBSs for climate change mitigation, including three new measures (creation of semi-natural wood pasture, semi-natural grassland and inter-tidal habitat), higher rates of peatland restoration, and additional low-carbon farming practices (Table 1). We then consider three variations on *NBS with biomass crops* to explore the impact of adding or removing specific measures: *NBS* excludes dedicated biomass crops, for which the mitigation benefits largely depend on Carbon Capture and Storage technology, which is unproven to work at scale; *NBS with organic farming* introduces organic farming (with no biomass crops) over 25% of the farmed area as one proven management practice that can increase biodiversity; and *NBS extra* deploys additional semi-natural grassland and wood pasture creation (with no biomass crops and no organic farming). Finally, to reflect future pathways in which limited land-use change occurs, *On Farm Measures (Organic)* and *On Farm Measures (Balanced)* are based on *NBS with organic farming* and *Balanced Pathway*, respectively, but use mainly on-farm mitigation measures. Following the CCC,[19] all scenarios include an increase in the area of urban land of 4% per 5-year period (replacing farmland adjacent to existing urban land), to accommodate a growing human population. All scenarios (except *Baseline*) include a package of low-carbon farming practices that reduce the GHG intensity of food production (i.e., emissions go down without reducing yields). Scenarios are numbered 0–8 in increasing order of climate change mitigation potential (see below).

We then estimate the consequences of each of the nine scenarios for (1) annual net GHG emissions from the land sector, (2) potential habitat availability for breeding birds, modeled as a function of future land use using data from the Breeding Bird Survey,[36] (3) food production, and (4) timber and biomass fuel production. We estimate GHG emissions for the land sector (Agriculture, Forestry, and Other Land Use), but do not consider downstream emissions associated with the transport, preparation, and decomposition of food. Separately, we also estimate emissions from imported livestock feed, agrochemical manufacture, and machinery manufacture and maintenance, although these sources are officially attributed to other sectors or countries. We focus on birds because they are a data-rich taxon but recognize that their response to land-use change is not necessarily representative of other taxa; our metric of habitat availability ignores pressures not directly linked to land use (such as climate, pollution, or disease), and assumes a constant relationship between land cover and local population density (i.e., constant habitat quality). Food, timber, and biomass fuel production are estimated from domestic sources only, excluding imports (with the exception of imported livestock feed used to rear domestic livestock). Finally, we do not distinguish between domestic food production for UK consumption versus export.

Land cover changes

By design, the extent of future land cover change varied substantially among scenarios (Figure 2), driving differences in modeled responses. Under *0: Baseline*, 98.1% of land remained unchanged between 2015 and 2050, with just 2.7% of initial arable and horticulture and 4.4% of initial improved grassland lost to urban expansion. In contrast, 26.9% of land area changed under the *NBS with biomass crops* scenario (22.5% when excluding expansion of agroforestry), which entailed reductions of 19.8% and 31.9% in the area of arable and horticulture and improved grassland, respectively (excluding changes to agroforestry), and a 60.1% increase in woodland cover (reaching 21.9% of total land cover, or 5,416 kha, by 2050). The area of wetland habitat (bog and fen, marsh and swamp) also increased (by 154% to reach 12.9% cover), as did wood pasture (reaching 1.3% cover), agroforestry (to 4.4% cover), and biomass crops (to 1.0% cover). While semi-natural grassland experienced a net decline under *6: NBS with biomass crops* (from 9.1% to

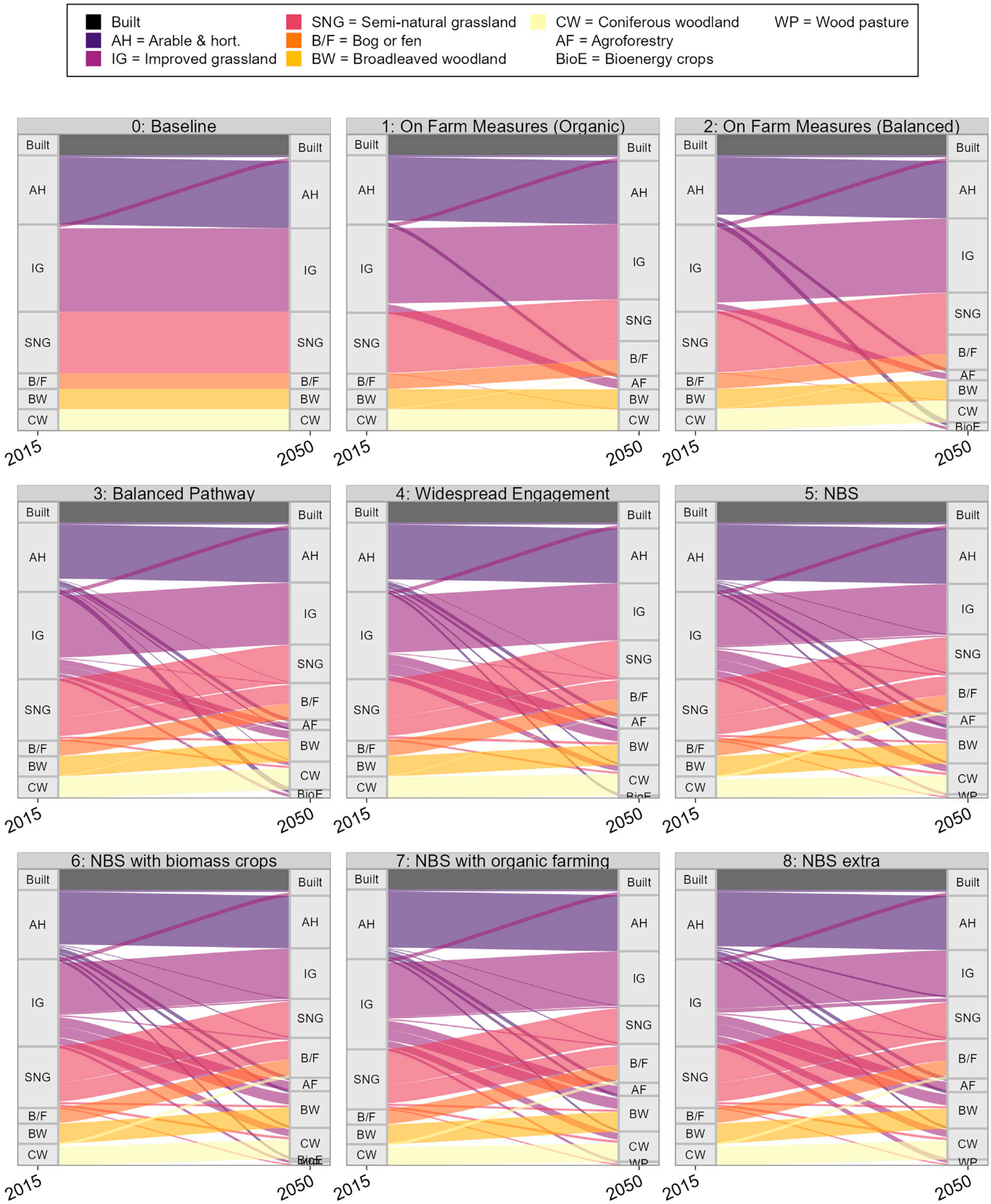

Figure 2. Land use/cover transitions between 2015 and 2050 under each scenario

Transitions involving coastal land, inland rock, freshwater or intertidal habitat, and those accounting for <0.1% of the total land area, are not shown. NBS, nature-based solutions.

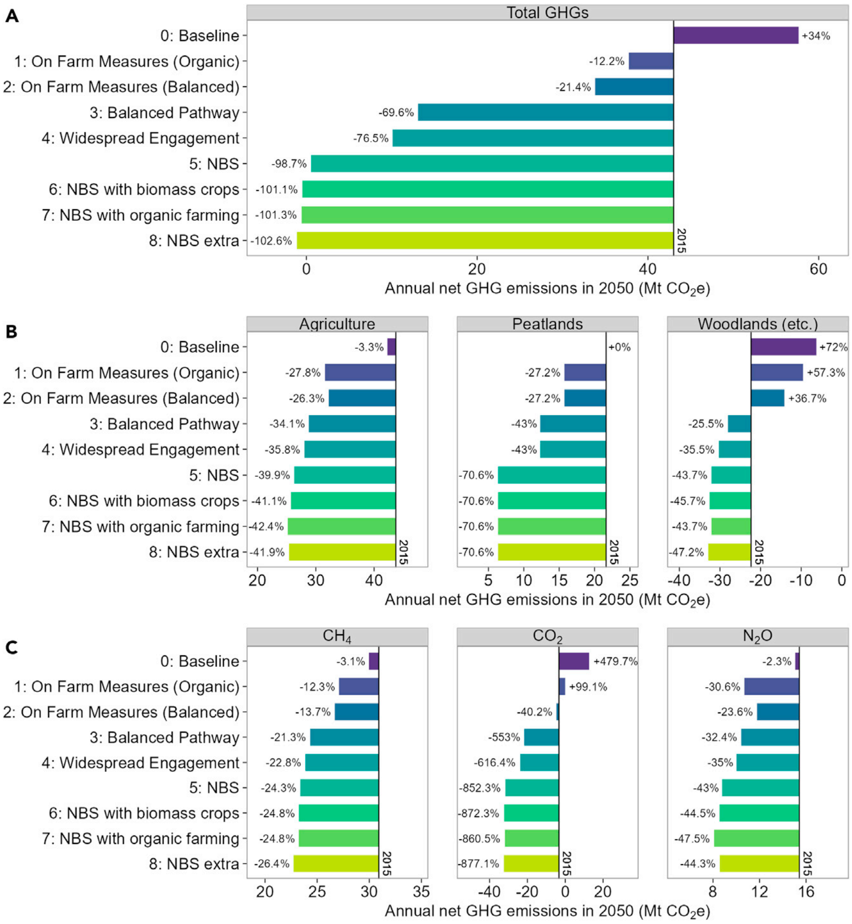

Figure 3. Annual net GHG emissions from the UK land sector in 2050 under each scenario

(A) combines emissions of all three GHGs for the whole land sector; (B) separates the land sector into agriculture, peatlands, and everything else; (C) separates GHG emissions into the three constituent gases. The vertical solid line shows estimated emissions in 2015. Excludes emissions from imported feed, agrochemical manufacture, and machinery manufacture and maintenance. NBS, nature-based solutions.

6.8% cover, primarily due to wetland habitat creation on peat soils), an additional 180 kha of semi-natural grassland were created on arable and horticulture and improved grassland (representing 0.7% of the total UK land area).

Net GHG emissions

Scenario *8: NBS extra* minimized net GHG emissions in 2050 (−1.1 Mt CO_2e, compared with 43.0 in 2015, and against 57.6 under *0: Baseline* in 2050; Figure 3A), and was one of four scenarios that delivered a net zero UK land sector (in 2050, remaining net negative for 21 years; Figure S1). Total cumulative avoided GHG emissions between 2015 and 2100 (compared with *0: Baseline*) reached −3,357 Mt CO_2e under *8: NBS extra* (annualized = 39.5 Mt CO_2e/year), equivalent to approximately 7 years' worth of current (2020) total UK territorial emissions.

Agriculture and peatlands remained a net source of GHG emissions under all scenarios, although the size of this source declined by up to 42% and 71%, respectively, under the most ambitious scenarios (Figure 3B). The remainder of the land sector (primarily woodlands and land-use change) provided a net sink under all scenarios. The size of this sink declined between 2015 and 2050 under scenarios 0–2, primarily as a result of sink saturation within existing woodland, but increased under scenarios 3–8, primarily as a result of woodland creation. Emissions of CH_4 declined by up to 26% and N_2O by up to 48% (Figure 3C). Net CO_2 emissions increased under scenario 0–1 but declined under scenarios 2–8.

In all scenarios, net annual emissions increased during the last third of the century as new woodlands created between 2020 and 2050 matured and provided a progressively diminishing annual sink (Figure S1). This conclusion was insensitive to assumptions regarding harvested wood product longevity (Note S1). For most scenarios, net annual emissions reached their lowest point around 2060.

Emissions from imported feed, agrochemical manufacture, and machinery manufacture and maintenance—sources that are typically attributed to other sectors or countries—were estimated at 16.4 Mt CO_2e in 2015. Including these additional sources increased net GHG emissions in 2050 under *8: NBS extra* to 9.4 Mt CO_2e, and meant that only three scenarios (*7: NBS with organic farming*, *8: NBS extra*, and *6: NBS with biomass crops*) achieved net zero (in 2057, 2058, and 2059, for 1, 2, and 3 years, respectively).

Our *3: Balanced Pathway* and *4: Widespread Engagement* scenarios delivered smaller emissions reductions than the corresponding CCC scenarios on which they are based.[19] This discrepancy is largely explained by CCC scenarios leveraging land-release levers (yield increases, dietary change, and reduced food waste) to deliver a large surplus of unused land (3,200 kha under *Widespread Engagement*). In contrast, our scenarios deliver no land surplus, and so involve smaller reductions in agricultural land and associated emissions.

Bird habitat index

We used two alternative modeling approaches to predict changes in bird habitat availability to gauge the sensitivity of our conclusions to analytical choices. Both models predicted similar changes in the bird habitat index across scenarios and species groups, and generated acceptable predictions for the 2015 population size of each species (Figure S2). We present the geometric mean and range of the predictions from the two models (Figure 4).

With the exception of *0: Baseline* (−2%; range = −3%, −1%), all scenarios saw a net positive change in the combined mean habitat index of 109 breeding bird species compared with 2015, ranging from +5% (+4%, +5%) under *2: On Farm Measures (Balanced)* to +18% (+18%, +19%) under *8: NBS extra*

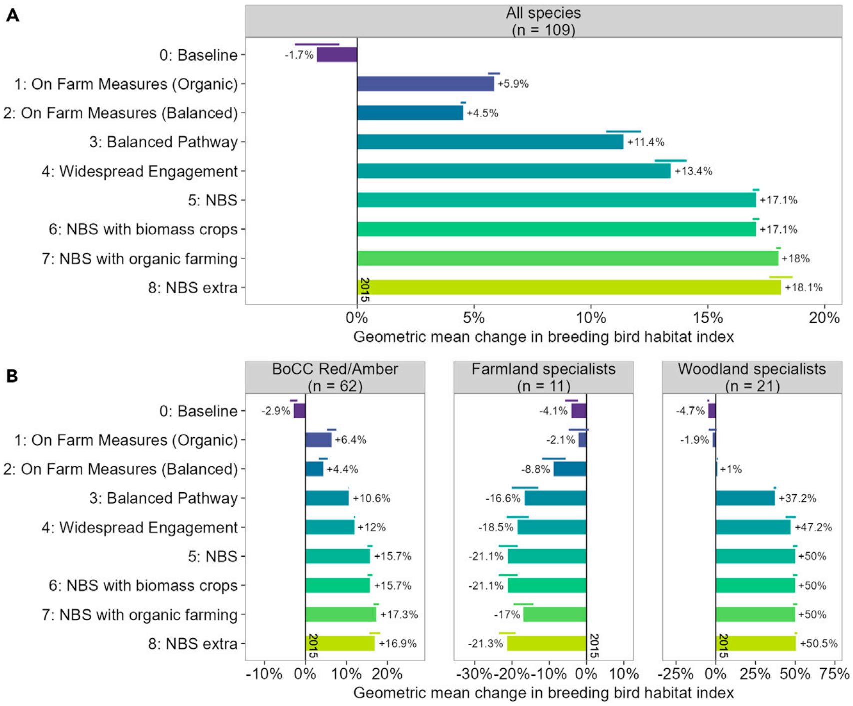

Figure 4. Geometric mean change in breeding bird habitat index under each scenario

(A) All species (n = 109), (B) Birds of Conservation Concern, farmland specialists, and woodland specialists. Bars show geometric mean of predictions from methods A and B; lines show range. NBS, nature-based solutions.

(Figure 4A). Results for Birds of Conservation Concern closely matched those for all species (−3%; −4%, −2% under *0: Baseline*, +4%; +3%, +5% under *2: On Farm Measures (Balanced)*, +17%; 16%, 18% under *8: NBS extra*), but species listed on farmland and woodland specialist indicator lists showed sharply contrasting patterns (Figure 4B). The habitat index for farmland specialists declined under all scenarios (by up to −21%; −24%, −19%), but least so under *1: On Farm Measures (Organic)* (−2%; −5%, +1%), while for farmland generalists the index declined under most scenarios (by up to −7%; −9%, −4%) but increased under *1: On Farm Measures (Organic)* (+4%; +1%, +6%). In contrast, the habitat index for woodland specialists and generalists increased under most scenarios (by up to +52%; +50%, 51%, and +25%; +24%, 26%, respectively).

Food production

All scenarios resulted in a reduction in food production by 2050 compared with the 2015 baseline (Figure 5). After *Baseline* (−3%, due to the loss of farmland to urban expansion), the scenario with the smallest reduction in food production was *2: On Farm Measures (Balanced)* (−14%). The scenario with the greatest reduction in food production was *7: NBS with organic farming* (−25%), followed by *6: NBS with biomass crops* (−23%; Figure 5).

Under most scenarios, production of animal-based products declined to a greater extent than production of crop-based products, primarily driven by reductions in grazing livestock (Figure 5B). Only scenarios *2: On Farm Measures (Balanced)* and *1: On Farm Measures (Organic)* saw a greater reduction in crop production than livestock production. Among crop-derived products, the composition of different food types varied little between scenarios.

The results presented in Figure 5 do not consider the impact of measures designed to close the gap between expected calorific supply and demand in 2050; these measures also influence the composition of food supply. For each scenario, multiple combinations of food waste reduction, feed crop substitution, and yield growth resulted in complete closure of the 2050 calorie gap (Note S2). Holding food waste reduction at 50% (the lower bound of the 2050 assumption used by the CCC[19]) and yield growth at 0% (assuming that any improvements are countered by the negative impacts of climate change) requires between 0% and 25% feed crop substitution, depending on the scenario. This value corresponds to the reduction in the fraction of each crop (on high-grade farmland) used for feed as opposed to direct human consumption; for a crop for which 60% of the harvest is used for feed and 40% for food, 25% feed crop substitution results in 45% going for feed and 55% for food. Feed crop substitution results in the replacement of pork, poultry, and eggs with plant-based products. As a fraction of total calorific supply (after waste), beef, lamb, and dairy declined by 7%–37%, and pork, poultry, and eggs by 3%–49%, depending on the scenario; conversely, plant-based products increased by 2%–14% (Note S2).

Ignoring the shading effect imposed by silvoarable agroforestry on the adjacent crop increased total calorific production by just 1.8%–2.2%, depending on the scenario. If silvoarable plantings were used for apple production rather than wood fuel (from poplar), we estimate that total calorific production would increase by another 0.3% (but total calorific production of fruit and vegetables would increase by ∼20%).

Timber and biomass fuel production

The total production of timber and biomass increased by up to 40% under scenarios that included expansion of dedicated biomass crops (*2: On Farm Measures (Balanced)* and *3: Balanced Pathway*), but decreased by 12%–15% under all other scenarios (Figure 6A). There was little variation in the production of timber between scenarios, which declined (by 12%–15%) in all scenarios due to changes in the age structure of existing forests. In contrast, fuel production increased (by up to 103%) under scenarios that included expansion of dedicated biomass crops, but decreased otherwise (Figure 6B).

Impacts on timber production are more apparent toward the end of the century, as woodlands created between 2020 and 2050 reach harvesting age. Both fuel and timber production increased under most scenarios, by up to 194% (*3: Balanced Pathway*) and 65% (all four *NBS* scenarios), respectively (Figure 6C).

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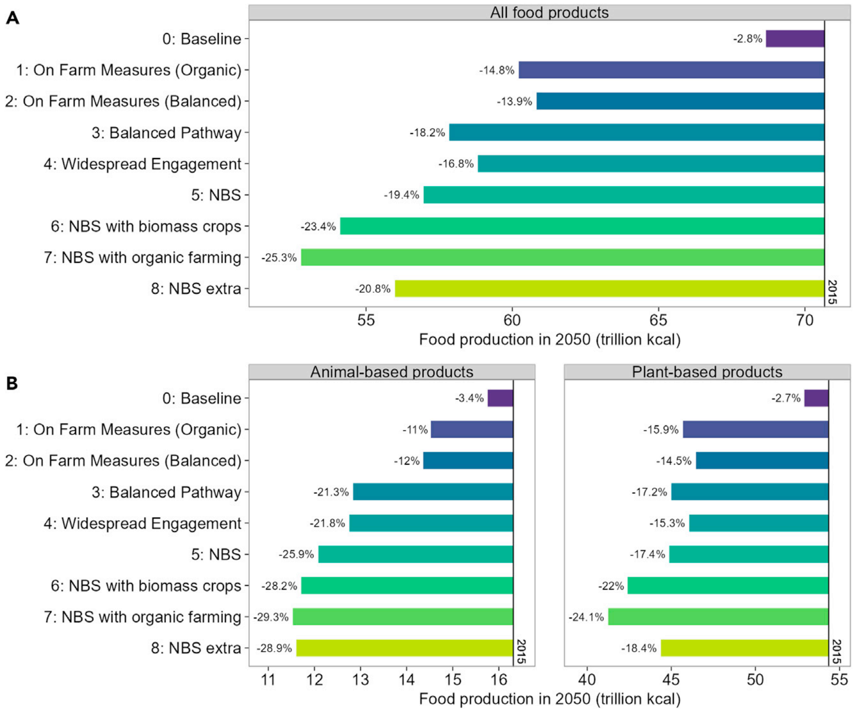

Figure 5. Food production in 2050 under each scenario

(A) Production across all food products; (B) separates animal-based products (beef, lamb, pork, poultry, eggs, and dairy) from plant-based products. NBS, nature-based solutions.

Trade-offs and synergies

Across scenarios, there was a strong trade-off between food production and both climate change mitigation and breeding bird habitat provision (Figures 7 and S3). The scenario that minimized net GHG emissions (*8: NBS extra*) resulted in the third largest reduction in food production and the largest increase in breeding bird habitat index. No scenario delivered strong reductions in GHG emissions (or large increases in the bird habitat index) without also resulting in a large reduction in food production. In contrast, neither timber nor biomass fuel production (2045–2054) strongly co-varied with any other outcome, although late-century production (2091–2100) correlated positively with the bird habitat index and climate change mitigation, and negatively with food production (Figures 7 and S3).

Modifying the constraints on new woodland

When woodland creation was avoided in areas supporting higher densities of woodland-sensitive conservation priority breeding wading birds, more arable land, but less semi-natural grassland and heathland, was subject to land-use change. Due to the greater loss of arable land, the reduction in food production compared with 2015 increased slightly from −16.8% to −18.8% under *4: Widespread Engagement* and from −19.4% to −20.6% under *5: NBS*; there was no effect on timber or biomass fuel production. Driven largely by avoided agricultural emissions, the reduction in net GHG emissions (2015–2050) increased from 77% to 79% under *4: Widespread Engagement* and from 99% to 100% under *5: NBS*. Predicted mean habitat index averaged across five farmland wading bird species (Eurasian curlew, common redshank, Eurasian oystercatcher, common snipe, and northern lapwing) was more positive compared with the default scenarios (+2% versus −3% for *4: Widespread Engagement*, +6% versus +2% for *NBS*).

DISCUSSION

Our scenario modeling exercise explores plausible ways in which UK land might change through efforts to reduce the land sector's contribution to climate change, illustrating how the contribution to multiple global challenges might be reconciled in a populous, high-income nation. The resulting scenarios are exploratory, illustrative, and non-exhaustive but highlight some opportunities, alongside challenges caused by land uses competing for a finite amount of space.

Is a net-zero UK land sector attainable?

Several of our scenarios are expected to come within ±1 Mt CO_2e of a net zero UK land sector by 2050, although only when emissions from fertilizer production and imported feed are excluded. For scenarios that deploy the full suite of mitigation measures at the highest ambition levels, up to 7 years' worth of current (2020) total UK territorial emissions are avoided over the course of the century. While this contribution of land-based mitigation may seem small, the UK has high total emissions given its land area (5-fold greater than the equivalent global average, from data in Roe et al.[16]).

Our projections beyond 2050 demonstrate that, if reached at all, net negative emissions are unlikely to be sustained in the long term without the deployment of additional mitigation measures, potentially requiring further land-use change. The expected increase in net emissions beyond ∼2060 is driven by a reduction in the size of the annual sink provided by new woodlands. This "sink saturation" occurs because the annual growth of woodland peaks after a few decades[37] and, for commercial forestry, biomass is eventually harvested through clear-felling. Although we assume that clear-felled forests are re-stocked, a substantial fraction of the original biomass decays to the atmosphere more quickly than it can be re-sequestered by growth during the second rotation. This reduction in the net sink provided by woodlands and forestry is relatively insensitive to assumptions about harvested wood product longevity (Note S1; see also Forster et al.[38]).

If the UK economy is to reach net zero by 2050, negative emissions are likely to be required to "net out" residual emissions. The land sector is the only sector currently capable of offering negative emissions at scale, so should ideally be providing a net sink by mid-century. For the land sector then, a target of net zero is arguably inadequate. Additional land-based GHG removal methods together have the potential to sequester 13 $MtCO_2$/year by 2050 (central estimate from the Department for Business Energy & Industrial Strategy[39]),

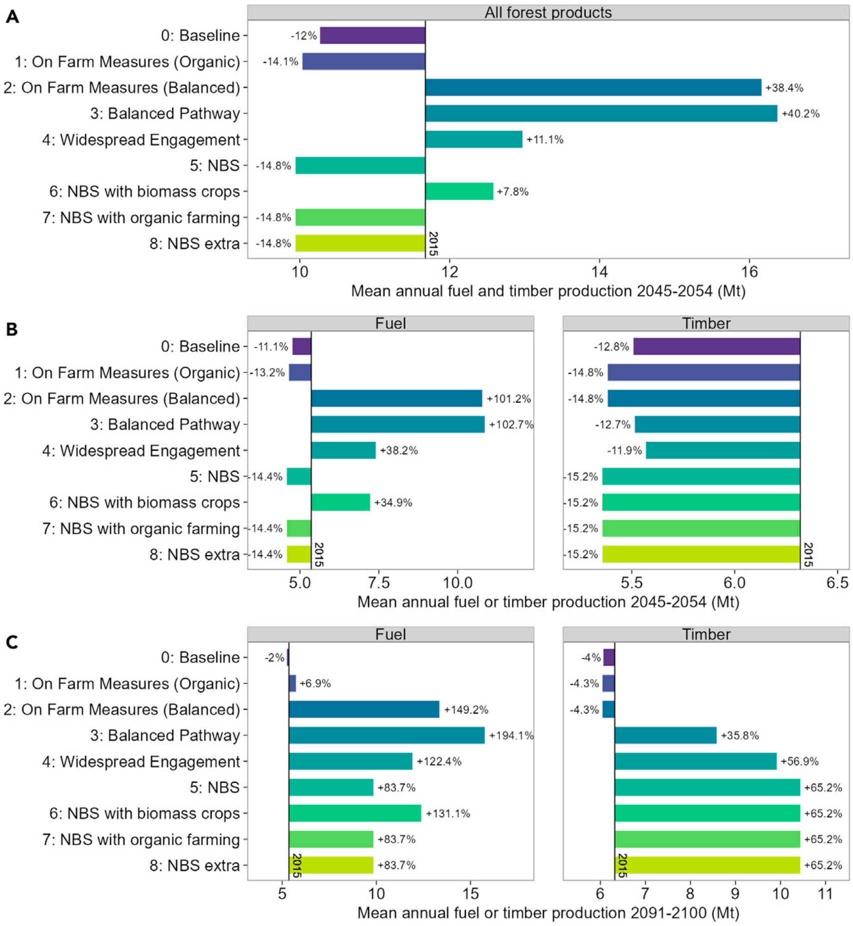

Figure 6. Mean annual production of biomass fuel and timber under each scenario
Fuel and timber production are combined in (A), and shown separately in (B) and (C), for periods 2045–2054 (A and B) and 2091–2100 (C). Solid vertical lines show mean annual production 2015–2020. NBS, nature-based solutions.

but each has some caveats. For example, uncertainties remain with respect to saturation and reversibility of soil carbon storage, and the efficacy of many regenerative farming methods is context dependent. Systematic field-scale testing of enhanced rock weathering is still lacking, and biochar requires land for biomass, which may compete with other measures. Engineered GHG removal methods such as Direct Air Carbon Capture and Storage or Bioenergy with Carbon Capture and Storage (BECCS) are still in their infancy, and depend on successful development of CO_2 transport and storage infrastructure.[40] Despite these limitations, many net zero plans and global Integrated Assessment Models are heavily reliant on such engineered removals to offset residual emissions from land and other sectors (e.g., the UK's Net Zero Strategy envisages 75–81 $MtCO_2$ of engineered removals by 2050[41]). The biomass fuel produced from forestry and dedicated crops under our scenarios could, if combined with BECCS, remove a modest 8.3–15.7 $MtCO_2$ per year by mid-century (assuming 50% carbon content and 90% capture rate). While these engineered removals may be crucial for eventually *reversing* climate change (by removing historic emissions), relying on them to *halt* climate heating (by offsetting ongoing emissions) is unproven, and we suggest that this reliance upon unproven technology is risky.[40] Instead, maximizing the nature-based mitigation potential of the land sector (which is substantial globally[6]) via already proven techniques can buy critical time while engineered removals are developed and eventually deployed at scale.[42]

Achieving multiple policy objectives from UK land

Our results highlight some striking relationships between modeled outcomes, with high-performing climate mitigation scenarios resulting in reduced food production but increased overall bird habitat availability.

The overall synergy between our climate and nature outcomes is encouraging, and in alignment with the "NBS" concept.[43] Global,[1,44] national,[23,30] and sub-national[31,45] studies have highlighted the potential for synergies between restoring biodiversity and reducing GHG emissions, although strategies that aim simply to maximize one outcome may result in sub-optimal performance for the other.[13,46] In contrast, Anderson et al.[14] report a trade-off between carbon density and priority species richness across Great Britain (GB). This conclusion is based on a negative spatial association between the *current value* of the two outcomes; our finding that future land-use change can improve both is not at odds with the fact that areas of high carbon density (e.g., blanket bogs) are often relatively species poor (although still of conservation importance).

Among bird species, different guilds showed divergent responses across scenario space, with woodland birds generally gaining habitat under high-performing climate mitigation scenarios and farmland birds losing habitat. This result mirrors the findings of Lamb et al.[23] Losses of farmland bird habitat could itself be mitigated by conservation interventions on remaining farmland[47,48] and, while such changes may incur a yield penalty, in some cases farmland conservation can act in synergy with food production, or at least incur a negligible yield cost.[49,50] Conversely, habitat loss might be exacerbated if measures such as woodland creation were to occur in ecologically unsuitable locations (e.g., peatlands, designated sites, mapped priority habitats), highlighting the importance of strategic spatial planning.

The projected decline in food production under high-performing climate mitigation scenarios is challenging, although unavoidable given that almost all the mitigation measures we considered result in a reduction in either the area or yield of

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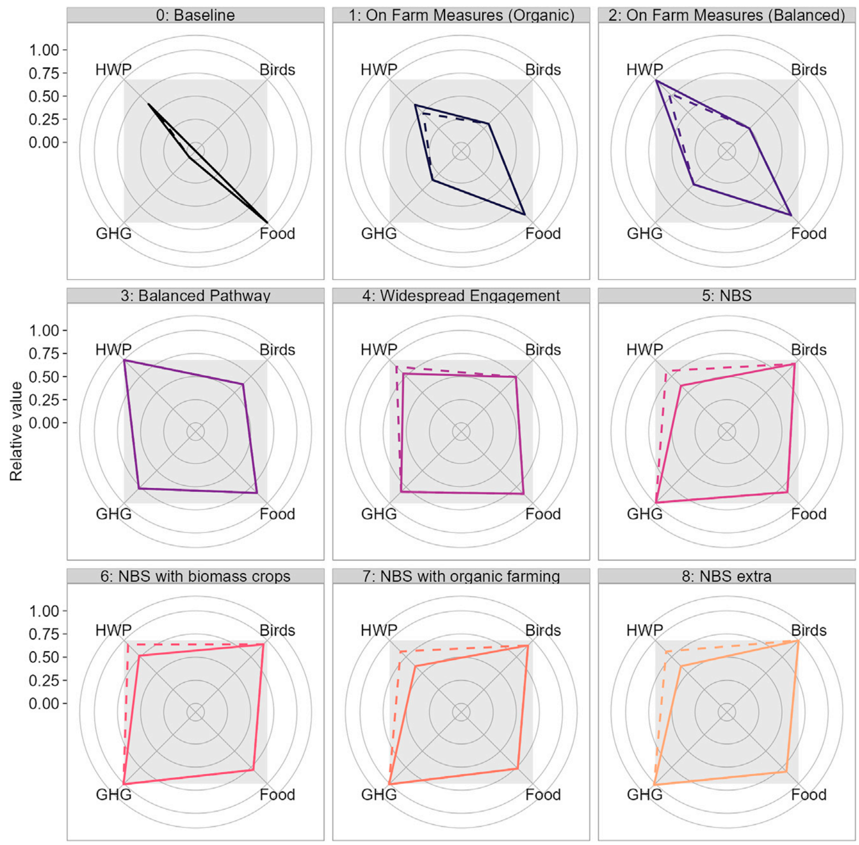

Figure 7. Summary of the performance of each scenario (panels) across four modeled outcomes

GHG, annual net GHG emissions in 2050; Food, 2050 calorific production; Birds, relative breeding bird habitat index; HWP (harvested wood products), annual fuel and timber production averaged 2045–2054 (solid line) and 2091–2100 (dashed line). Each outcome is expressed relative to its maximum value across scenarios. The GHG axis has been transformed ($-x+1$) such that high values correspond to low net emissions. The gray polygon illustrates a "preferred" outcome where all variables achieve their maximum (or minimum for GHG emissions) value across scenario. NBS, nature-based solutions.

farmland. The magnitude of the decline in food production may be surprising given the findings of the National Food Strategy[51] that "the least productive 20% of land [in England] produces only 3% of our calories." While we avoid creation of woodland, wood pasture, and semi-natural grassland on the most productive and versatile farmland, and prioritize habitat creation on lower-grade farmland first, measures such as urban expansion, lowland peatland restoration, hedge creation, organic farming, and agroforestry do impact more productive farmland. The challenge of reducing GHG emissions while maintaining food production is well established[52,53] and emphasizes the importance of demand-side solutions such as dietary change or waste reduction.[54,55]

In Note S2 we explore different combinations of food waste reduction, feed crop substitution and yield growth needed to fill the 2050 "calorie gap" under each scenario, increasing domestic food supply (after waste) in proportion to expected population growth. In all cases, our scenarios demand a partial shift away from both red and white meat, toward a more plant-based diet. Large-scale dietary change is challenging, although between 2008 and 2019, self-reported daily per capita meat consumption in the UK declined by 17%.[56] On food waste, the UK has a target to cut avoidable waste by 50% between 2007 and 2030, and has already reported a 27% reduction between 2007 and 2018.[57] Potential for future yield growth is less clear, with the yields of many crops seeing little change across much of Europe in recent years[58]; nonetheless, observed cereal yields are ~30% lower than modeled estimates of potential maximum yields, suggesting room for improvement.[59] While strategies deploying yield growth delivered smaller reductions in GHG emissions and smaller increases in the bird habitat index, shunning yield growth entirely means more food waste reduction or feed crop substitution are required to close the calorie gap (Note S2). A key challenge will be ensuring that yield growth does not exacerbate environmental externalities (i.e., sustainable intensification[60]). "Alternative" production systems such as peri-urban horticulture, vertical indoor production, precision fermentation, or cultured meat could also help close the calorie gap, in some cases with limited land-use competition.[61,62] In the long term, transitioning to food production methods that involve considerably less land may present the ultimate solution to the global challenges addressed in this study,[63] although we focus on the immediate need for "transitional" solutions. These actions are imperative if we are to avoid the offshoring of forgone calorie production from the UK having an impact on biodiversity and carbon in countries from which food would need to be imported.[27,64,65]

An additional challenge for UK agriculture arises from climate change impacts, although these are hard to forecast due to the competing effects of a longer growing season, CO_2 fertilization, and adverse weather impacts.[66,67] Unpredictable non-linear changes in climate might result in more fundamental regime shifts, such as the cessation of arable cropping across large parts of the country due to water limitation (changes to irrigation notwithstanding).[68] Climate impacts, while difficult to quantify, arguably present a greater threat to food production than the mitigation measures deployed in our most ambitious scenarios. We also ignore the potential benefits to food production of nature restoration, although these benefits are likely to be greatest for interventions that improve soil function or create fine-scale habitat features aimed at beneficial invertebrates in and around productive areas,[49] as opposed to the larger-scale measures deployed in our scenarios.

Due to the lag between woodland establishment and eventual harvest, mid-century timber production did not strongly co-vary with any other outcome. However, high-performing climate change mitigation scenarios saw increases in timber and biomass production by late century. UK wood product consumption is currently heavily import dependent, with the overseas land footprint of UK timber, pulp, and paper consumption equivalent to more than half the UK's land area (13.3 Mha[69]). Meeting near- to medium-term wood demand without further increasing import dependence will require better management and utilization of existing UK forests and forest products. This could deliver additional climate change mitigation by slowing the decay of harvested wood products[39] and substituting carbon-intensive building materials within the construction sector[38] (but see Leturcq[70]). Demand-side solutions include recycling and re-using existing wood products and avoiding the burning of biomass, which could otherwise be put to longer-term use.[38]

Assumptions, uncertainties, validation, and calibration

Where possible, we modeled changes in GHG emissions from the land sector using official UK GHG Inventory accounting methodology. Estimates of uncertainty for GHG inventories are themselves "highly uncertain" (https://ghgprotocol.org/sites/default/files/2023-03/ghg-uncertainty.pdf), making it challenging to estimate the confidence in our predictions. The availability of independent figures with which to validate our predictions is limited due to between-study differences in scenario assumptions, but in Table S1 we compare estimated 2050 emissions from forestry, peatlands, hedges, agroforestry, bioenergy crops, and urban expansion between our study and the corresponding CCC (2020) scenarios,[71] finding good correspondence. We also find excellent correspondence for mid- and late-century timber production, although not for biomass fuel production (Table S2).

Our model of bird habitat availability does not capture the impacts of population drivers such as climate change, disease, or pollution, so validating our predictions against observed population trends is not appropriate. However, the fact that two independent model formulations yielded similar predictions gives us confidence in our approach, and both models generated acceptable predictions for the 2015 population size of each species (Figure S2, although our main interest is in relative differences between scenarios).

Our food production model was calibrated to exactly reproduce reported 2015 production across different categories of agricultural produce. Predictions of future production are sensitive to the way in which this baseline production is subsequently apportioned across UK farmland (according to land cover, region, and agricultural land capability), as this determines the loss of production that occurs following land-use change. Encouragingly, when we used our model to calculate the fraction of calories derived from the least-productive 20% of English land, we derived a similar figure (2%) to that reported in the National Food Strategy (3%),[51] giving confidence in our approach. While there is limited data on the impact of silvoarable agroforestry on crop yields, our overall conclusions are relatively insensitive to this source of uncertainty. Future changes in per-hectare productivity are uncertain and are not captured by our model.

Conclusions

Our scenarios illustrate the potential of NBSs to deliver both climate change mitigation and biodiversity conservation in a populous, high-income nation, although we identify risks to farmland birds. Our most ambitious scenarios deliver a net zero land sector by 2050 without resorting to unproven engineered removals such as BECCS. However, the lack of "spare" mitigation potential highlights the critical role of immediate, aggressive, and permanent emissions reductions across all sectors to achieve economy-wide net zero. Achieving multiple objectives from finite land requires strategic, spatially explicit policies spanning the land and food systems. Land-use decisions are complex, however, and cannot be informed by ecological and biophysical modeling alone; understanding the social, cultural, political, and economic dimensions is critical, including the distribution of benefits and disbenefits among stakeholders with potentially competing interests,[72] both within the UK and overseas. We stress that business as usual is not an option, given the UK land sector's continuing contribution to climate change.

EXPERIMENTAL PROCEDURES

Resource availability

Lead contact

Further information and requests for resources should be directed to and will be fulfilled by the lead contact, Tom Finch (tom.finch@rspb.org.uk).

Materials availability

This study did not generate new unique materials.

Data and code availability

Where permitted by external licensing agreements, all data will be shared by the lead contact upon request after publication (Table S3 lists the availability of all third-party data sources). Original code, in addition to summary results, have been deposited in a Zenodo repository (Zenodo: https://doi.org/10.5281/zenodo.8269104).

Summary of spatially explicit scenario design

Each scenario is represented by a 25-m land cover raster for each 5-year period between 2015 and 2050. The 2015 baseline raster is derived from the UKCEH Land Cover Map 2015 (LCM2015[35]), which maps 21 land cover categories at 25-m resolution across the UK.

For scenarios *4: Widespread Engagement* and *5: NBS*, we developed alternative spatial realizations of the mitigation measures and ambition levels described in Table 1 and Figure 1 to account for an additional potential constraint. Specifically, we avoided woodland creation, agroforestry, wood pasture, and biomass crops on areas identified as important for farmland-breeding wading birds (Eurasian curlew *Numenius arquata*, common redshank *Tringa totanus*, Eurasian oystercatcher *Haematopus ostralegus*, common snipe *Gallinago gallinago*, and northern lapwing *Vanellus vanellus*), for which woodland expansion represents a particularly acute threat.[21] For these two alternatives, we still prioritized measures such as woodland creation on lower-carbon mineral soils, recognizing that net climate mitigation may be lower due to carbon pool change,[33] but also permitted measures on non-peat organomineral soils.

For the purposes of estimating annual GHG emissions and food supply, and timber and biomass fuel production, we used linear interpolation to convert five-yearly land cover rasters into tabular data representing annual land cover (and land cover change) areas.

Underlying spatial data

Land cover

The foundation of our scenarios is the UKCEH Land Cover Map (LCM2015[35]), which maps 21 land cover categories at 25-m (0.0625-ha) resolution. We excluded raster pixels with no land cover data and those classed as saltwater, leaving 211.2 million pixels in England (13.2 Mha), 127.8 in Scotland (8.0 Mha), 33.9 in Wales (2.1 Mha), and 22.7 in Northern Ireland (1.4 Mha). For simplicity, we aggregated the 21 land cover categories to 16 (Table S4).

Next, we identified the NUTS1 government office region (Nomenclature of Territorial Units for Statistics, representing Scotland, Wales, Northern Ireland,

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and nine English regions) of each pixel. We rasterized a shapefile of NUTS1 regions[73] to the land cover raster extent using the *fasterize* function in R,[74] which identifies the polygon value over the centroid of each raster pixel. For pixels with land cover data but no overlapping NUTS1 data, we identified the NUTS1 value of the nearest pixel (Euclidean distance). London and South East England were combined into a single region. Land cover data were then separated into four country-specific rasters.

We updated LCM2015 data using country-specific spatial datasets covering woodlands and other priority habitats. We made the following updates:

(1) First, we assigned to broadleaved woodland any pixels classed as non-PAWS (Plantations on Ancient Woodland Sites) ancient woodland (England, Wales), as woodland priority habitat (Northern Ireland) or as non-pinewood native woodland (Scotland). We assigned to coniferous woodland any pixels classed as PAWS woodland (England, Wales, Scotland) or as native pinewood (Scotland).[75–78]
(2) Next, we assigned to broadleaved woodland any pixels classed as broadleaved woodland, mixed mainly broadleaved woodland, coppice, or coppice with standards (England, Wales, Scotland) or as broadleaved or mixed woodland (Northern Ireland). We assigned to coniferous woodland any pixels classed as coniferous woodland, mainly coniferous mixed woodland or windthrow (England, Wales, Scotland) or as coniferous woodland (Northern Ireland).[79,80]
(3) Finally, we assigned to broadleaved woodland, coniferous woodland, acid grassland, neutral grassland, calcareous grassland, bog, fen, heathland, saltmarsh, or coastal any pixels classed as such in national priority habitat inventories.[81–86]

For pixels not covered by these national habitat or woodland inventories, we retained the original land cover data from LCM2015.

All spatial data (described below) were rasterized to the same resolution and extent as the land cover data using the *fasterize* function in R.[74] For pixels with missing data, we assigned the value of the nearest pixel with non-missing data.

Urban expansion

To account for potential changes in the area of built land, we followed the CCC[19] which projects an increase of 23.9% between 2019 and 2050 (equivalent to a 3.86% increase per 5-year period). We made the simplifying assumption that built land replaces arable and horticulture and improved grassland only. We ranked existing arable and horticulture and improved grassland pixels in ascending order of distance to existing built pixels (randomizing the rank of pixels with equal distance). We then converted arable and horticulture and improved grassland pixels (excluding those on peat soil or earmarked for intertidal habitat creation—see below) to built land, increasing the total area of built land in each country by 3.86% of the 2015 area in each 5-year period between 2020 and 2050.

These five-yearly land cover rasters formed the basis of all scenarios.

Land capability for agriculture

To map spatial variation in agricultural land capability, we used the Agricultural Land Classification (ALC) (or the equivalent in Scotland: Land Capability for Agriculture), which grades the quality of land for agricultural use according to the versatility and suitability for growing different crops (Table S5).[87–90] In England, where the ALC maps do not separate grades 3a and 3b, we performed a manual split according to two criteria. First, grade 3 pixels with a slope >7° (calculated from SRTMG3 Digital Elevation Model[91]) were assigned to grade 3b. The remaining grade 3 pixels were split according to soil type, using the Harmonised World Soils Database.[92] Soil types which contained more grade 1/2 land than grade 4/5 land in England were identified as "high grade," and assigned to grade 3a. Pixels with slope $\leq 7°$ and not on high grade soil were assigned to grade 3b.

Opportunities for peatland restoration

We derived a shapefile of peat extent from Bradfer-Lawrence et al.[17] Pixels earmarked as opportunities for intertidal habitat creation (see below) were classified as non-peat, as were those with incompatible land covers (inland rock, saltmarsh, coastal, and built).

To identify which peat pixels were likely to be restored to bog versus fen, we made a simplifying assumption that peat within one vertical meter of a river channel would revert to fen. To calculate vertical distance to river channels, we first filled sinks in the 3 arc second resolution SRTMG3 Digital Elevation Model[91] using the SAGA tool "Fill sinks (Wang & Liu)" in QGIS (minimum slope = 0.01), then identified river channels using the "r.stream.extract" tool (minimum flow accumulation = 50). After rasterizing the resulting channel lines to the same resolution as the DEM, we calculated the vertical distance from each non-channel pixel to the nearest channel pixel, and used a simple threshold to delineate potential fen (≤ 1 m) from bog (>1 m). This approach may result in degraded raised bogs being misclassified as candidates for restoration to fen, especially where peat wastage has lowered the ground level with respect to nearby river channels.

For the purposes of prioritizing peatland restoration, we used shapefiles of national designated sites (*Sites* and *Areas of Special Scientific Interest* and *National Nature Reserves*) and Natura2000 sites (*Special Protection Areas* and *Special Areas of Conservation*) from the European Environment Agency.[93]

Forest yield class and opportunities for woodland creation

To approximate spatial variation in tree yield class (and thus carbon sequestration rate) we used Forest Research's Ecological Site Classification (ESC) tool.[94] We first mapped 24 climatic strata across the UK, based on WorldClim v.2 30-s resolution maps of mean annual temperature and total annual rainfall averaged 1970–2000.[95] We identified eight strata based on mean annual temperature (rounding to the nearest whole degree Celsius and combining categories 2°C–4°C and 11°C–12°C), then divided each temperature stratum in three according to the 33.3% and 66.6% quantiles of annual precipitation. For each stratum, we sampled 20 random coordinates, and extracted expected climatic yield class for silver birch (*Betula pendula*), Scots pine (*Pinus sylvestris*), and Sitka spruce (*Picea sitchensis*) from the ESC online tool for each coordinate. We then calculated the mean yield class of each species in each stratum and translated these yield values back to the mapped climatic strata.

To identify opportunities for woodland creation, we used an existing UK-wide map that excludes deep peat soils, mapped priority habitats, and designated sites.[17] Any remaining pixels classed as peat, as well as those earmarked for intertidal habitat creation (see below), were excluded. We also excluded pixels classed as coniferous woodland, broadleaved woodland, inland rock, freshwater, saltmarsh, coastal, and built, as well as those with a predicted silver birch yield class <2 as these areas were assumed unsuitable for woodland creation. Finally, we excluded pixels identified as grade 1, 2, or 3a (3.1) farmland (Table S5), representing the best and most versatile farmland.

Opportunities for woodland establishment through natural colonization were identified as pixels within 100 m of existing broadleaved woodland or native pinewood consistent with dispersal parameters of relevant trees species described by, e.g., Gerber et al.[96,97]

To identify organo-mineral soils at risk of soil carbon loss following woodland creation[10,33] we used the Harmonised World Soil Database (HWSD) raster data,[92] calculating the mean %topsoil organic carbon and %topsoil clay across all soil units within each mapping unit, weighted according to the share occupied by each soil unit. Following Bol et al.,[98] mapping units with %organic carbon $\geq 6\%$ and %clay <50% were classified as organo-mineral, and excluded from woodland creation. For mapping units with %clay <50%, the %organic carbon threshold was set to $0.05 \times \%$ clay + 3.5.

To identify sites of populations of ground-nesting birds species that avoid breeding in areas close to woodland,[21] we acquired tetrad-level (2×2 km) predictions of relative abundance for curlew, snipe, redshank, lapwing, and oystercatcher from the Bird Atlas 2007–11.[99] For each species, we excluded tetrads whose cumulative abundance summed to less than 10% of total abundance, and then excluded the bottom 33.3rd percentile of remaining tetrads. Remaining tetrads were deemed "high-strata," which we combined across all five species, encompassing between 85% (lapwing) and 97% (redshank) of total predicted relative abundance. For two supplementary scenarios, all pixels in high strata tetrads were excluded from woodland creation.

In Scotland, we mapped the current and potential extent of native pinewood using the Caledonian Pinewood Inventory.[100] Pixels classed as coniferous woodland, which overlapped "Caledonian Pinewood" polygons, were treated as native pinewood. Other pixels overlapping Caledonian Pinewood or "Regeneration Zone" polygons were treated as candidates for restoration to native pinewood, as opposed to native broadleaved or managed coniferous woodland.

Opportunities for semi-natural grassland creation

For arable and horticulture and improved grassland pixels we estimated the most likely semi-natural grassland type based on underlying soil. We cross-tabulated HWSD mapping units[92] with the 2015 land cover data, then identified the most frequent semi-natural grassland type (acid, neutral, or calcareous grassland) within each soil mapping unit. We excluded pixels identified as grade 1, 2, or 3a (3.1) farmland.

Opportunities for intertidal habitat creation

We obtained shapefiles of sites identified by a previous opportunity mapping exercise[101] as being potentially suitable for intertidal habitat creation. The shapefile was tidied to remove polygons representing entire grid squares, leaving only polygons representing mapped site boundaries, which were classified by Miles and Richardson[101] as either "priority" or "non-priority" opportunities. We then rasterized these polygons, excluding pixels already classified as saltwater, saltmarsh, or coastal, to leave 8,835 ha of priority opportunities (across 49 sites) and 19,977 ha of non-priority opportunities (across 245 sites).

Hedge length

We calculated hedge length as a parameter for modeling breeding bird densities (see below) using data from the Woody Linear Features Framework[102] for GB and the Copernicus Small Woody Features[103] for Northern Ireland. The former is a vector layer, which we rasterized to the same resolution of the land cover data (i.e., identifying 25-m pixels that are intersected by a mapped hedge). The latter is a 5-m resolution raster layer, which we aggregated 5-fold (ignoring pixels identified as small woods) and reprojected to the same extent as the land cover data. We estimated km-per-pixel separately for GB and Northern Ireland by dividing the total number of hedge pixels in each region by the reported total hedge length in each region (705,000 km for GB in 2007,[104] 115,000 for Northern Ireland in 2007, extrapolating the reported figure for 1998 according to the proportional change in GB hedge length 1998–2007[105]).

Mitigation measures

The following sections describe each climate change mitigation measure and associated ambition levels in detail. Table S6 describes the relevance of each mitigation measure for food production, GHG emissions, timber and biomass fuel production, and bird populations.

Intertidal habitat creation

Intertidal habitat creation involves the wholesale replacement of terrestrial land covers with saltmarsh habitat (treated as a combination of saltmarsh and mudflat for the purposes of estimating carbon sequestration; see below). This mitigation measure does not compete with other land cover changes, as we exclude pixels identified for intertidal habitat creation from the opportunity area of other measures. We considered two ambition levels for intertidal habitat creation:

(1) *Low* ambition: no intertidal habitat creation. This scenario implies an annual loss of 105 ha of saltmarsh due to sea level rise and coastal squeeze,[101] although we do not explicitly model this loss (except for the purposes of calculating greenhouse gas balance of saltmarsh);
(2) *High* ambition: create 0.48 kha of new intertidal habitat per year (2050 total = 0.0144 Mha), so that ~50% of the 28,865 ha of restoration opportunities identified by Miles and Richardson[101] are realized between 2020 and 2050. This level of ambition implies a ~10-fold increase in saltmarsh creation rates, which are currently ~45 ha per year.[101]

We converted pixels identified as opportunities for intertidal habitat creation to saltmarsh on a per-site basis. We assigned each unique restoration site to a 5-year period at random (priority sites were restored before non-priority sites), ensuring a maximum increase in the area of new intertidal habitat of 2.4 kha in each 5-year period (= 0.48 kha per year). The small area of built land lost to intertidal habitat creation (165 ha, equivalent to 0.01% of the total area of built land in 2015) was not compensated for, but we recorded the area of coniferous and broadleaved woodland lost to intertidal habitat creation in each 5-year period, adding this to woodland creation targets (see below).

Peatland restoration

Peatland restoration is restricted to pixels identified as peat soil (see above) and involves a combination of both land cover change and condition improvement to reduce GHG emissions from degraded or managed peatlands. Peatland restoration does not compete with other land cover changes, which we exclude from pixels identified as peat. We considered four ambition levels for peatland restoration, which result in increasing rates of restoration and rewetting of different categories of degraded peat:

(1) *Low* ambition: no peatland restoration;
(2) *Medium* ambition: all extensive grassland and modified bog restored by 2045; 20% of forestry on peat restored by 2035; all peat extraction sites restored by 2030; 10% of lowland cropland converted to paludiculture and 30% under water table management by 2050. Based on CCC[19] "*Headwinds*" pathway;
(3) *High* ambition: all extensive grassland and modified bog restored by 2045; 20% of forestry on peat restored by 2035; all peat extraction sites restored by 2030; 50% of intensive grassland and 25% of lowland cropland restored by 2050; 15% of lowland cropland converted to paludiculture and 35% under water table management by 2050. Based on CCC[19] "*Balanced Net Zero Pathway*" and "*Widespread Engagement*" pathways;
(4) *High+* ambition: all extensive grassland and modified bog restored by 2045; 100% of forestry on peat restored by 2035; all peat extraction sites restored by 2030; 75% of intensive grassland and 50% of lowland cropland restored by 2050; 50% of lowland cropland and 25% of intensive grassland converted to paludiculture.

For the purposes of identifying different categories of degraded peat (and for estimating GHG emissions) we paired each land cover category with one or more peatland condition categories.[106] For land covers arable and horticulture, coniferous woodland, and improved grassland, we assumed a straight pairing with peatland condition categories "cropland," "forest," and "intensive grassland," respectively (Table S7). We assumed that semi-natural grassland (acid, neutral, or calcareous) corresponded to peatland condition categories "extensive grassland," "drained grass-dominated modified bog," and "undrained grass-dominated modified bog." These condition categories are not mapped, so we assumed that each semi-natural grassland pixel represented a combination of the three, reflecting the country-specific proportional area of each activity category in 2013, calculated from Evans et al.[106] Similarly, we assumed that each heathland pixel represented a combination of "drained heather dominated modified bog" and "undrained heather dominated modified bog," and that fen represented a country-specific combination of "rewetted fen," and "near-natural fen." Finally, we assumed that each bog pixel represented a country-specific combination of "rewetted bog," "near-natural bog," "drained eroded modified bog," "undrained eroded modified bog," and peat extraction (domestic and industrial). The small number of peat pixels associated with freshwater land cover (0.4% of the total peat area) were not altered in our scenarios and did not contribute to peat-associated GHG emissions. Broadleaved woodland currently on peat (and coniferous woodland identified as native pinewood) was also left unaltered, under the assumption that no planting of broadleaved trees on peat has occurred, and so any broadleaved woodland on peat soil is likely to be semi-natural.[107]

Peatland restoration involves the conversion of arable and horticulture, coniferous woodland, improved, acid, calcareous, and neutral grassland and heathland pixels to either bog, fen, or paludiculture, as well as changes in the condition of bog (from modified and extraction sites to rewetted) and arable and horticulture (through water table management). We implemented land cover changes in 100-ha (i.e., 1,600 25-m pixel) units, but restricted changes within each 100-ha unit to 25-m pixels on peat of the relevant land cover. For each 100-ha unit, we first calculated the proportional area under national or European designation, the modal ALC value (see above) and the distance to the nearest 100-ha unit containing bog or fen in 2015. Based on 5-year restoration targets for each peat condition category, we converted peat pixels to bog or fen in descending order of protected area cover (highest first), then descending order of ALC (highest first, reflecting lower agricultural capability), then ascending order of distance to the nearest semi-natural peatland (closest first). For each 25-m pixel we distinguished potential bog from fen according to vertical distance to the nearest channel (see above). Bog and fen restoration targets (up to 2050) were fulfilled prior to allocating cropland and improved grassland to paludiculture.

Woodland creation

Woodland creation involves a change in land cover to either broadleaved woodland or coniferous woodland. Within each 5-year period, the creation of new broadleaved woodland or native pinewood took priority over exotic coniferous woodland. We fulfilled woodland creation targets to 2050 before deploying other measures. We considered three ambition levels for woodland creation, varying in both the amount and type of new woodland:

(1) *Low* ambition: no new woodland;
(2) *Medium* ambition: 30 kha/year by 2025, rising to 50 kha/year by 2035 (2050 total = 1.19 Mha). Based on CCC[19] *Balanced Net Zero Pathway*;
(3) *High* ambition: 42 kha/year by 2025, rising to 70 kha/year by 2035 (2050 total = 1.69 Mha). Based on CCC[19] *Widespread Engagement* pathway.

We created new woodland in 25-ha (i.e., 400 25-m pixels) units, but within each 25-ha unit woodland creation was only permitted on 25-m pixels identified as opportunities for woodland creation (see above). Units of 25 ha were converted to either native woodland (including native pinewood, see above) or managed coniferous woodland, with the former making up 66.6% of the target area in each country in each 5-year period. For each 25-ha unit we first calculated the area of existing coniferous and broadleaved woodland, the distance to the nearest 25-ha unit containing coniferous and broadleaved woodland, and the modal agricultural land class. We ranked each 25-ha unit separately for coniferous and broadleaved woodland in descending order of ALC (highest first, reflecting lower agricultural capability), then descending order of area of existing woodland of the same type (highest first), then ascending order of distance to the nearest existing woodland of the same type (closest first). The limited opportunities for native pinewood restoration were prioritized for the first 5-year period.

In all cases, headline UK woodland creation targets were allocated to each country in proportion to the area of woodland opportunity (see above) after excluding land earmarked for urban expansion. Based on 5-year restoration targets for each woodland type (supplemented by the area of each woodland type lost to inter-tidal habitat creation and peatland restoration in each 5-year period), we then converted pixels within each 25-ha unit to woodland. Following the CCC,[71] we inflated woodland creation rates by 15% to accommodate for open ground within new woodlands.

Wood pasture creation

Wood pasture creation involves introducing trees to existing areas of acid grassland, calcareous grassland, neutral grassland, or heathland. After fulfilling 2050 woodland creation targets, wood pasture creation took priority over semi-natural grassland and agroforestry. We considered three ambition levels for wood pasture:

(1) *Low* ambition: no wood pasture;
(2) *Medium* ambition: 16.7 kha/year (2050 total = 0.5 Mha);
(3) *High* ambition: 25 kha/year (2050 total = 0.83 Mha).

We converted land to wood pasture in 25-ha units, assuming that opportunities for new wood pasture followed the same constraints as for new woodland (excluding grades 1, 2, and 3a (3.1) farmland). We ranked each 25-ha unit in ascending order of distance to the nearest existing woodland. The total area of new wood pasture was allocated to each country in proportion to the area suitable for wood pasture (according to current land cover and woodland opportunity). Based on 5-year creation targets we assigned pixels in each 25-ha unit to wood pasture.

We defined wood pasture as having 30% cover of woody vegetation (leaving 70% as semi-natural grassland), divided equally between semi-natural broadleaved woodland and scrub.

Semi-natural grassland creation

Semi-natural grassland creation involves the conversion of arable and horticulture and improved grassland to acid, neutral, or calcareous grassland. After fulfilling 2050 woodland and wood pasture creation targets, semi-natural grassland creation took priority over agroforestry. We considered three ambition levels:

(1) *Low* ambition: no new semi-natural grassland;
(2) *Medium* ambition: 6 kha/year (2050 total = 0.18 Mha);
(3) *High* ambition: 15 kha/year (2050 total = 0.45 Mha).

We converted arable and horticulture and improved grassland to semi-natural grassland in 25-ha units, avoiding peat pixels and those earmarked for inter-tidal habitat creation. For each 25-ha unit we first calculated the area of mineral soil, the distance to the nearest 25-ha unit containing semi-natural grassland, and to the nearest 25-ha unit containing woodland. We ranked each 25-ha unit in ascending order of mineral soil cover (lowest first, to minimize competition with woodland creation), then ascending order of distance to the nearest existing semi-natural grassland (closest first), then in descending order of distance to the nearest existing woodland (furthest first). The total area of new semi-natural grassland was allocated to each country in proportion to the current area suitable for new semi-natural grassland (including both mineral and organo-mineral soils). Based on 5-year restoration targets, we then converted arable and horticulture and improved grassland pixels within each 25-ha unit to semi-natural grassland. We determined the grassland type of each 25-ha unit according to the current distribution of acid grassland, neutral grassland and calcareous grassland with respect to broad soil types (see above).

Agroforestry

Agroforestry involves planting trees in existing areas of arable and horticulture (silvoarable) and improved grassland (silvopasture). Agroforestry was permitted only on land which had not already been used for woodland, wood pasture or semi-natural grassland creation. We consider three ambition levels:

(1) *Low* ambition: no agroforestry;
(2) *Medium* ambition: 16.7 kha/year silvopasture and 11 kha/year silvoarable by 2050 (total = 0.88 Mha). Based on CCC[19] *Balanced Net Zero Pathway*;
(3) *High* ambition: 25 kha/year silvopasture and 11 kha/year silvoarable by 2050 (total = 1.08 Mha). Based on CCC[19] *Widespread Engagement* pathway.

We converted land to agroforestry in 25-ha units, assigning silvopasture to improved grassland pixels and silvoarable to arable and horticulture pixels. We assumed that opportunities for new agroforestry followed the same constraints as for new woodland, and therefore only considered agroforestry on pixels identified as opportunities for woodland creation (see above), although we permitted agroforestry on grades 1, 2, and 3a (3.1) farmland. We ranked each 25-ha unit in ascending order of distance to the nearest existing woodland. The total area of new agroforestry was allocated to each country in proportion to the area suitable for each agroforestry type (according to current land cover and woodland opportunity). Based on 5-year agroforestry creation targets we assigned available pixels in each 25-ha unit to agroforestry.

Following Thompson et al.,[71] we defined silvopastoral agroforestry as broadleaved woodland planted at 400 trees/ha (occupying 14% of the grassland area) and silvoarable agroforestry as broadleaved woodland planted at 42 trees/ha (occupying 6.7% of the cropland area).

Hedge creation

Hedge creation involves the creation of rows of woody shrubs at field boundaries. We consider three ambition levels for hedge creation:

(1) *Low* ambition: no new hedge creation;
(2) *Medium* ambition: 40% increase in hedge length by 2050 (total = 328,000 km). Based on CCC[19] *Balanced Net Zero Pathway* and *Widespread Engagement* pathways;
(3) *High* ambition: 50% increase in hedge length by 2050 (total = 410,000 km).

We chose not to explicitly model the spatial distribution of new hedges, but we estimated the current distribution of hedges across the UK, and applied increases in hedge length pro rata.

Biomass crops

Biomass crops are permanent crops that replace arable and horticulture or improved grassland, producing material for combustion for energy (potentially with subsequent CO_2 storage). We restricted biomass crops to land which, by 2050, had not been subject to other land cover changes, and considered three levels of ambition:

(1) *Low* ambition: no new bioenergy crops;
(2) *Medium* ambition: 10 kha/year of *Miscanthus* by 2031 (2050 total = 0.24 Mha). Based on CCC[19] *Widespread Engagement* pathway;
(3) *High* ambition: 10 kha/year each of *Miscanthus*, short-rotation coppice (SRC) and short-rotation forestry (SRF) by 2031 (2050 total = 0.71 Mha). Based on CCC[19] *Balanced Net Zero Pathway*.

We replaced arable and horticulture and improved grassland pixels with biomass crops in 9-ha units (i.e., 144 25 m-pixels, approximating individual fields). Following the CCC,[19] we restricted *Miscanthus* and SRC to arable and horticulture and SRF to improved grassland. We assumed that opportunities for biomass crops followed the same constraints as for new woodland, and therefore only considered these measures on pixels identified as opportunities for woodland creation (see above). The total area of biomass crops was allocated to each country in proportion to the current suitable area for *Miscanthus*, SRC, and SRF. Otherwise, biomass crops were introduced to 9-ha units of arable and horticulture and improved grassland at random, in line with five-yearly targets.

Organic farming

Organic farming implies changes in land management at the farm scale, resulting in reduced artificial inputs, reduced yields, and modified crop rotations. We restricted new organic farming to land which, by 2035, had not been subject to other land cover changes, but did not protect new organic farming from land cover change after 2035. We considered two ambition levels for organic farming:

(1) *Low* ambition: no additional organic farming;
(2) *High* ambition: 25% of farmland converted to organic by 2035, reflecting the EU's Farm to Fork Strategy.[34]

We converted arable and horticulture and improved grassland pixels (including those already converted to silvoarable or silvopastoral agroforestry) to organic farming in 100-ha units (i.e., 1,600 25-m pixels, approximating entire farms; Defra[108]) at random. In each 5-year period, the area of organic land in each country increased linearly, such that 25% of all arable and horticulture and improved grassland were organic by 2035.

Low-carbon farming

Low-carbon farming involves a combination of farming practices and technological innovations that reduce agricultural emissions without reducing the area of farmed land. We considered four ambition levels:

(1) *Low* ambition: no additional low-carbon farming practices;
(2) *Medium* ambition: 50%–75% uptake of behavioral measures and 50%–78% uptake of innovative measures; electrification of agricultural machinery; plus a 20% and 10% improvement in nitrogen use efficiency on cropland and grassland, respectively. Based on CCC[19] *Balanced Net Zero Pathway*;
(3) *High* ambition: 60%–80% uptake of behavioral measures and 50%–78% uptake of innovative measures; electrification of agricultural machinery; plus a 30% and 10% improvement in nitrogen use efficiency on cropland and grassland, respectively. Based on CCC[19] *Widespread Engagement* pathway;
(4) *High+* ambition: 60%–80% uptake of behavioral measures and 50%–78% uptake of innovative measures; electrification of agricultural machinery; a 30% and 10% improvement in nitrogen use efficiency on cropland and grassland, respectively; plus additional "sustainable" practices.

Behavioral and innovative practices, their respective rollout periods and uptake under each ambition level, and their abatement potential are defined according to the CCC[19] and Eory et al.[109]

Cover crops

We assume that non-cash cover crops are integrated into crop rotations, with uptake increasing from 30% (*Baseline*) to 78% (*Medium*) or 80% (*High* and *High+*) over 5 years from 2022.[110] We apply this measure to 34% of the area of spring crops (representing sandy or silty soils where cover crops are most applicable). We conservatively assume that cover crops have no impact on long-term carbon sequestration but reduce leaching by 45%.

Integrating grass leys into rotation

We assume that temporary grassland is re-integrated into a 4-year arable rotation, with the measure being rolled out to 5% (*Medium*, *High*, and *High+*) of temporary grassland over 10 years from 2022. We assume that this sequesters an additional 0.202 t CO_2e ha^{-1} $year^{-1}$ in these areas.

Grass-legume mixtures

We assume that uptake of legume-grass mixtures across temporary and improved grassland increases from 26% (*Baseline*) to 75% (*Medium*) or 80% (*High* and *High+*) over 10 years from 2022. Legume-grass mixtures reduce fertilizer requirements and therefore reduce associated emissions; we assume that nitrogen requirements fall from 99 to 50 kg N ha^{-1} on temporary grassland and 50 to 0 kg N ha^{-1} on permanent grassland.[107,110]

Covering slurry tanks

We assume that the use of impermeable covers for liquid manure storage increases from 0% to 75% (*Medium*) or 80% (*High* and *High+*) over 5 years from 2022. Impermeable covers reduce airflow over the surface of slurry tanks and thus reduce N_2O and NH_3 emissions by 100% and 80%, respectively, for the two gasses, and the CH_4 conversion factor by 47%.

High sugar content grasses

We assume that current uptake of high sugar content grasses (HSGs) is 9%, rising to 75% (*Medium*) or 80% (*High* and *High+*) over 5 years from 2022. Compared with the baseline diet, we assume that HSGs reduce N excretion per liter of milk by 9%, thus reducing emissions from manure management.

Precision feeding

We assume that uptake of precision feeding increases from 0% to 75% (*Medium*, *High*, and *High+*) over 5 years from 2022, reducing the nitrogen and volatile solid excretion of dairy cattle and pigs by 2%.

3NOP as feed additive

3-Nitrooxypropanol (3NOP) is a feed additive that reduces the production of enteric CH_4. We assume that uptake of 3NOP increases from 0% to 50% (*Medium*, *High*, and *High+*) over 10 years from 2025, reducing enteric CH_4 emissions by 30% in dairy cattle and 20% in beef cattle.

Improved livestock breeding (genomics)

We assume that the use of genomics to improve livestock breeding delivers a 0.15% annual reduction in cattle CH_4 emissions factors from 2030 to 2050, and that uptake rises from 0% to 60% from 2030 over 10 years.

Improved livestock breeding (genetic modification)

We assume that the use of genetic modification delivers an additional 0.4% annual reduction in cattle CH_4 emissions factors from 2040 to 2050, with uptake rising from 0% to 50% from 2040 over 10 years.

Electrification

According to the CCC,[19] stationary and mobile combustion of fossil fuels in the agricultural sector emitted 4.2 Mt CO_2*e* in 2015. The 6th Carbon budget pathways include reductions in emissions from farm machinery through measures such as electrification and hydrogen fuel. While the details underlying these emissions reductions are unclear, reported abatement from agricultural machinery by 2050 under BNZP is equivalent to 41% of 2018 emissions. We therefore apply this level of emissions reduction across *Medium*, *High*, and *High+* ambition levels, assuming the 41% total is achieved via a linear reduction in emissions between 2022 and 2050.

Improved nitrogen use efficiency

This measure was not included as a low-carbon farming measure by the CCC,[19] with nitrogen use efficiency (NUE) instead leveraged to deliver yield growth without additional fertilizer requirements. However, because our core scenarios include no yield growth, we deploy improved NUE as a low-carbon farming practice, defining the magnitude of efficiency improvements of 20% under *Medium* ambition and 30% under *High* and *High+* ambition. We assume that NUE improvements accrue linearly between 2022 and 2050, resulting in

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reductions in nitrogen application rates with no impact on yields. For scenarios that involve yield growth, we assume no change in nitrogen application rates.

To identify additional low-carbon practices to promote under the *High+* ambition level, broadly categorized as sustainable farming practices, we referred to reviews of climate change mitigation measures in agriculture.[110–112] We identified the following measures that are likely to deliver additional abatement beyond the measures already described, both of which are rolled out over 5 years from 2022.

Nitrogen-fixing crops in rotation

Following Lampkin et al.[112] we assume that 50% of arable land sees legume crops (beans and peas) expand to occupy 15% of the arable rotation (excluding temporary grassland). Legume crops require no nitrogen inputs and are assumed to reduce the nitrogen requirements of the subsequent crop by 20 kg ha^{-1}. This measure is applied to non-organic arable land, as organic rotations already include 15% cover of legumes.

Loosening compacted soils

Following Eory et al.,[110] we assume that 20% of arable land and grassland suffers from compaction, and that 100% of this land is loosened through measures such as subsoiling, topsoil cultivation, and shallow spiking. We assume that loosening compacted soils reduces direct soil N_2O emissions by 40%.[110]

Estimating food production

Our food production evaluation model estimates total agricultural production (in kcal) from UK land for each scenario-year. We include all UK crop and livestock production, including livestock that are fed crops grown on land overseas. We do not distinguish between products grown for export versus domestic consumption. Our starting point for estimating food supply from each scenario is the reported harvested production of 9 crop products, 35 horticultural products, and 6 livestock products for the UK in the baseline period (averaged 2014–2016; Table S8). Our methodology allocates this production to 25-m pixels to estimate the food supply consequences of changing the land cover of any pixel.

Arable and horticulture

Crop areas

Pixels classed as arable and horticulture were assumed to represent a NUTS1-region-specific combination of 16 arable crops (spring- and winter-sown wheat, barley, oats, oilseed rape, peas, and field beans; sugar beet, potatoes, and other cereals; and grain maize), fodder crops (whole maize and other fodder crops), horticulture, and temporary grassland. We established the proportional area of these 20 crop categories in each NUTS region based on national agricultural census statistics.[113,114]

Temporary grassland is identified as arable and horticulture in LCM2015, but there were mismatches between the reported area of crops and temporary grassland in each NUTS1 region and the total area of arable and horticulture LCM2015 pixels. In most regions, this mismatch implied that some temporary grassland had been misidentified as improved grassland in LCM2015. We therefore calculated, for each NUTS1 region, the proportion of temporary grassland on arable and horticulture pixels (ranging from 3% in the East of England to 49% in Northern Ireland, mean = 13.7%) and on improved grassland pixels (ranging from 0% in the East of England and East Midlands to 16% in Wales, mean = 7%).

We adjusted crop areas within each NUTS1-ALC combination, under the assumption that all potatoes, sugar beet, and horticultural crops are restricted to grade 1, 2, 3a, or 3b land, with the small area of arable land on lower-grade land supporting cereals, legumes, and fodder crops only.

Crop yields

We estimated the typical yield of each of the 16 arable crops (excluding horticulture, fodder crops, and temporary grassland, which are dealt with separately), averaged over the period 2014–2016. Yields were adjusted so that the sum of the product of NUTS1-level crop areas and yield reproduced the total reported production of each crop product (from Table S8, excluding grain maize for which no estimates of total production were available), maintaining the relative yield of spring- and winter-sown varieties. Yields of horticultural crops (per hectare of horticultural land) were calculated by dividing national production by the total reported area of fruit, vegetables, and protected horticulture. Fodder crops and temporary grassland are dealt with below.

Next, we estimated the relative yield of each ALC category following Table 2 in Moxey et al.,[115] which presents estimated winter wheat yield in classes 2, 3.1, 3.2, 4.1, and 4.2 for different fertilizer regimes. We calculated the mean yield across fertilizer regimes, expressed relative to class 3.1. Classes 4.1 and 4.2 were combined. For class 1 we assumed an increase in yield relative to class 2 equivalent to half the difference between classes 2 and 3.1, and for class 5 we assumed a decrease in yield relative to class 4 equivalent to half the difference between classes 3.1 and 4. In Scotland, sub-divisions of classes 4 and 5 were treated equally, and classes 6 and 7 were treated as class 5.

Crop production

We then calculated the baseline area of *arable and horticulture* pixels in each NUTS1 region and each ALC, then estimated the production of each crop in each NUTS1-ALC combination as the product of (1) national crop-specific yield, (2) NUTS1- and ALC-specific reported area of each crop, (3) ALC-specific relative yield, and (4) ALC-specific proportional area in each NUTS1 region. To derive per-pixel production, we divided the production of each crop in each NUTS1-ALC combination by the number of pixels in each NUTS1-ALC combination.

Finally, according to Food and Agriculture Organization of the United Nations,[116] we estimated the proportion of each crop used for food (as opposed to feed, seed, or other industrial uses, but ignoring "losses," which are accounted for later) (Table S9). We assumed that lower-grade land (4 or worse) produced feed-grade crops only, and so increased the fraction used for food on grade 1, 2, 3a, or 3b land to match the overall feed/food ratio from baseline production. The fraction used for feed was converted to livestock products as described below.

Grass

Baseline grass production

We assume that grazing livestock products (beef, lamb, and dairy) are fed on home-grown grass, supplemented with home-grown fodder crops and feed crops. We assume that any surplus home-grown feed crops are used to produce non-grazing livestock products (pork, poultry, and eggs), with unmet feed requirements met through imports. We first estimated baseline grass production as the quantity of grass required per ton of ruminant livestock product (from Table S10), multiplied by total production of each ruminant livestock product (from Table S8).

Grass yields

Next, we attributed baseline grass production (33.6 Mt dry matter) to grass-producing pixels as a function of land cover. The highest-yielding grasslands are improved temporary and permanent grasslands, to which we assigned relative yields of 1 and 0.64, respectively.[117] In each NUTS1 region we calculated a weighted relative grass yield for *arable and horticulture* (according to the regional ratio of temporary grassland to non-grass crops; see above) and *improved grassland* (according to the regional ratio of temporary to permanent grassland; see above). Semi-natural habitats also produce grass, although at lower yields. We estimated relative grass yields for semi-natural habitats compared with permanent grassland according to the recommended stocking density in each habitat compared with improved grassland.[118]

We then allocated total baseline grass production in proportion to each pixels' relative yield.

Livestock distribution

We assumed that the production of ruminant-derived livestock products (i.e., beef, dairy, and lamb) was limited by domestic grass production, with each grass-producing pixel supporting a NUTS1-region-specific combination of sheep, beef cattle, and dairy cattle according to agricultural census data.[108,119–121] For each scenario, we calculated total grass production across all grass-producing pixels, then calculated total ruminant production by multiplying grass production by the requirements of each livestock type, maintaining the proportional abundance of each grazing livestock type in each NUTS1 region, but assuming that dairy cattle only consumed grass from *improved grassland* and *arable and horticulture* pixels.

Fodder crops

Next, we estimated the quantity of fodder crops required to supplement the diet of cattle and sheep reared on the available grass, according to the fodder requirements described in Table S10. We calculated fodder crop yields assuming that baseline fodder crop requirements were exactly met from the reported area of fodder crops and fodder maize assumed to represent 95%

of the total area of maize, with the remaining 5% producing maize grain for feed, dealt with below.[113,114] Fodder crop production was attributed to *arable and horticultural* pixels as described above, accounting for regional variation in the area of fodder crops relative to other arable crops, and spatial variation in yield associated with Agricultural Land Class.

For scenarios in which fodder crop production exceeded requirements (due to reductions in grazing livestock numbers), we converted surplus fodder crop hectares to the region-specific composition of other arable crops.

Feed crops

Domestic feed crop production was estimated according to the fraction of each crop product used for feed (Table S9). The total mass of feed crops was converted into energetic units (MJ) according to the metabolizable energy content of each feed crop averaged across ruminants, pigs, and poultry (from https://feedipedia.org/).

We first estimated the remaining feed requirements of cattle and sheep reared on the available grass (from Table S10). Any surplus feed was then converted into pork, poultry meat, and eggs, according to the feed requirements in Table S10 and maintaining the baseline ratio of these three livestock products. To match current levels of non-ruminant production, we assumed that any remaining feed requirements were met through imported feed. We scaled future feed imports in proportion to domestic feed crop production, but constrained non-ruminant production to avoid increases in feed imports relative to 2015.

Total food supply

Total food supply was calculated by converting the total tonnage of food crop products and livestock products into calories using nutritional information from the USDA Nutrient Database for crops and horticultural products (additionally using information from De Laurentiis et al.[122] to exclude the inedible portion of each horticultural product), and from Wilkinson[123] for livestock products. While a focus on calories alone is likely to miss important components of a nutritionally sufficient diet,[124] the FAO's standard definition of undernourishment is based on minimum dietary energy requirements, and when we estimated food supply under each scenario in protein terms, 2050 protein production (Mt) was almost perfectly correlated with 2050 calorie values (Pearson's correlation; $r = 0.999$, $df = 7$, $p < 0.001$).

Impact of mitigation measures on food supply

For agroforestry and hedge creation we eliminated grass and crop production from the direct footprint of trees, hedges, and scrub (6.7% for silvoarable, 16% for silvopasture, and 30% for wood pasture). For silvoarable agroforestry, we applied an additional yield penalty of 29% to the crops grown between trees to account for possible shading effects[125–127]; recognizing the uncertainty of such shading effects, we compared results with and without this yield penalty. We assumed that hedges were 1.5 m wide, implying the removal of 0.15 ha of agricultural land per km of new hedge (this represents a compromise; some hedges will be established on existing margins and unproductive areas, whereas others will be wider than 1.5 m after accounting for management buffers). We assume no additional impact of trees or woody vegetation on grass or livestock yields.

For organic farming, we applied yield penalties specific to each arable crop and grassland type. For crops, relative organic yields were taken from de Ponti et al.'s meta-analysis.[128] We also specified a minimum area of 15% temporary grassland and 10% legumes (beans and peas) on arable land, reducing the fraction of other crops pro rata. Grass yields were reduced by 10% following Muller et al.[129]

We assumed that low-carbon farming practices had no impact on yields, except where changes in crop rotations influenced the ratio of different crop outputs. For newly created semi-natural grassland and wood pasture grazed by cattle and sheep, we assumed a shift to 100% grass-based diets. We replaced the feed and fodder requirements (from Table S10) with grass such that total metabolizable energy was unchanged. This implies a reduction in livestock production per unit area.

Closing the calorie gap

To explore options for closing the gap between UK-wide food production and expected demand, we considered three levers: reducing food waste, substituting feed crops for food crops, and yield growth. We first calculated expected calorific demand in 2050 by multiplying 2015 production by 1.125, assuming a 1:1 relationship between human population growth and demand for domestically produced food (medium variant population projections for UK from the United Nations[130]). Next, we converted food production to food supply by subtracting the product-specific fraction of each food type that is wasted in the post-harvest food supply chain, following Lamb et al.[25] (Table S11). We then identified combinations of food waste reduction (reducing the fraction of each crop type wasted by up to 70%, in line with the CCC[19]), feed crop substitution (reducing the fraction of each crop used for feed, and using the same crop for direct human consumption), and yield growth (increasing yields on non-organic arable and horticulture and improved grassland by up to 34%, in line with the CCC[19]), which closed the gap between supply and demand to the nearest 1% (Note S2).

Estimating timber and biomass fuel production

For the purposes of estimating both timber and biomass fuel production and GHG emissions (see below) we modeled the growth dynamics of new and existing woodland using data from the Woodland Carbon Code Biomass Carbon Lookup Table v.2.0.[131] We used linear interpolation to derive annual estimates from five-yearly values of carbon (t/ha) in standing biomass and debris, as well as biomass removed through periodic thinning. We converted carbon to total biomass (oven dry tons [odt]) by multiplying by 2.[132] For coniferous woodland we used biomass curves representing thinned Sitka spruce (1.7-m spacing), except for pixels identified as Caledonian pinewood (Scotland only), for which we used curves representing unthinned Scots pine (2-m spacing). For broadleaved woodland (80% unthinned, 20% thinned) and SRF (thinned) we used biomass curves representing sycamore/ash/birch (*Acer pseudoplatanus*/*Fraxinus excelsior*/*Betula* sp.: 2.5-m spacing). We treated new agroforestry and wood pasture as unthinned sycamore/ash/birch woodland, but scaled down per-hectare biomass values to reflect the lower areal footprint of trees compared with woodland. Note that, for sycamore/ash/birch, Woodland Carbon Code curves were available only for unthinned stands, so we extrapolated values for thinned stands from unthinned values according to the relationship between thinned/unthinned Oak.

Next, to account for climate-associated spatial variation in woodland growth rates, we estimated the likely yield class of each 25-m pixel representing new and existing woodland and new wood pasture using the Ecological Site Classification tool[94] for Sitka spruce, Scots pine, or silver birch (see above). For silvoarable agroforestry, silvopastoral agroforestry, and SRF we assumed a fixed yield class of 12, 6, and 12, respectively (following Thompson et al.[71]).

For existing woodland, the growth stage of each individual pixel is unknown, so we weighted age-specific biomass values according to the reported overall age structure of coniferous and broadleaved woodland.[133] For new woodland created within 100 m of existing woodland, we assumed that establishment was achieved through natural colonization rather than planting[96,134]; we conservatively delayed biomass growth curves by 5 years and reduced the estimated yield class by 2 units compared with woodland established through planting.[131]

We assumed that 100% of coniferous woodland (except Caledonian pinewood) and 20% of broadleaved woodland was subject to rotational clear-felling,[71] on a species- and yield-class-specific rotation length, calculated by rounding the year of maximum annual increment[135] to the nearest decade (range = 40–70 years). We assumed rotation lengths of 35 years for silvoarable agroforestry, 44 years for silvopastoral agroforestry, and 26 years for SRF,[71] and assumed no clear-felling of wood pasture. We assumed a 3-year fallow period before re-stocking,[17] and treated "deforestation" (i.e., loss of woodland to either inter-tidal habitat creation or peatland restoration) as clear-felling (with biomass harvested and included in production estimates) without re-stocking. We assumed that clear-felling resulted in the removal of the stem, branches, and foliage, estimating the fraction of total standing biomass (above- and below-ground) contained within these components for coniferous and broadleaved trees separately.[132] Combining this with biomass removed through periodic thinning, we thus calculated the total quantity of biomass (odt) removed in each scenario-year.

We also estimated the fraction of removed biomass used for timber (including board) and fuel, separately for biomass arising from thinning and clear-felling, and for coniferous and broadleaved woodlands[132] (Table S12). We stress, however, that these fractions are representative averages, and will vary depending on future growing conditions and silvicultural practices. For agroforestry and SRF we assumed that 100% of biomass removed was used for fuel. We also calculated biomass harvested from perennial energy crops on cropland (assuming 12 odt per hectare per year for both *Miscanthus* and SRC, following Thompson et al.[71]).

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Estimating net GHG emissions

Our net GHG emission evaluation model estimates the total net flux of GHGs from the land sector in each scenario-year. We express emissions in terms of CO_2e, using IPCC AR4 100-year global warming potential values (GWP_{100} = 25 for CH_4, 295 for N_2O[136]).

Agricultural emissions

To estimate changes in agricultural emissions, we scaled reported 2015 emissions[107] in proportion to changes in the underlying driver of each emission source under each scenario (for example, enteric CH_4 emissions were scaled in proportion to changes in gross energy consumed by different livestock types). We include emissions from the agricultural sector as defined by IPCC guidance (dominated by N_2O and CH_4), as well as emissions from agricultural activities (including fertilizer and pesticide manufacture, machinery manufacture and maintenance, and feed imports), which are typically accounted for in other sectors or countries.[25]

Livestock emissions

Baseline (2015) emissions of CH_4 from enteric fermentation and CH_4 and N_2O from manure management were taken directly from the UK GHG Inventory[107] for each livestock type (beef cattle, dairy cattle, sheep, pigs, poultry, and other livestock; Table S13). For future scenarios, we scaled enteric CH_4 emissions in proportion to the change (relative to baseline) in gross energy consumed by each livestock type, following.[25] CH_4 emissions and direct N_2O emissions from manure management were similarly scaled in proportion to changes in the metabolizable energy and crude protein consumed by each livestock type, respectively. We assumed no change in future emissions from deer, horses and goats (representing a small fraction of baseline emissions; Table S13).

Emissions were allocated to each of the four UK countries in proportion to the gross energy, metabolizable energy or crude protein consumed by each livestock type in each country, assuming that ruminants are distributed in proportion to grass production and non-ruminants in proportion to feed production.

Baseline indirect N_2O emissions from manure management (1,365 t N_2O via volatilization and 30 t N_2O via leaching in 2015, for all UK livestock production) were taken from the same source (Brown et al.[107]) but are not reported separately for different livestock types. Emissions factors from the UK GHG Inventory were therefore used to approximate the fraction of indirect N_2O emissions attributable to each livestock type, according to the fraction of manure handled by different manure management systems. As above, for future scenarios we varied indirect N_2O emissions in proportion to the change (relative to baseline) in crude protein consumed by each livestock type, following Lamb et al.,[25] and allocated emissions to each of the four UK countries in proportion to the crude protein consumed by each livestock type in each country.

N_2O emissions from nitrogen applied to agricultural soils

Baseline direct N_2O emissions from agricultural soils were taken directly from Brown et al.,[107] separately for inorganic nitrogen, animal manure, and sewage sludge applied to soils, and urine and dung deposited on soils, as well as from crop residues (Table S14). Baseline indirect N_2O emissions from agricultural soils (1,693 t N_2O via volatilization and 5,713 t N_2O via leaching) were taken from the same source (Brown et al.[107]), but are not reported separately for different nitrogen sources. Emissions factors from the UK GHG Inventory were therefore used to approximate the fraction of indirect N_2O emissions attributable to each source, accounting for the reported quantity of nitrogen applied.

Direct and indirect N_2O emissions from manure applied to soils were attributed to different livestock types according to reported emissions factors, the fraction of manure handled in different manure management systems, and the fraction of nitrogen lost under each system.[107] Similarly, direct and indirect N_2O emissions from urine and dung deposited on soils were attributed to different livestock types according to reported emissions factors and the fraction of manure handled in the "pasture/range/paddock" system.[107] Under future scenarios, we scaled N_2O emissions (and allocated emissions to each country) in proportion to the change in crude protein consumed by each livestock type.

Baseline national average inorganic nitrogen application rates (kg/ha) for each crop type (including spring and winter varieties, and temporary and permanent grassland) were derived from Brown et al.[107] Total inorganic nitrogen use was calculated for each scenario by summing the area of each crop type multiplied by the crop-specific application rate. We then scaled baseline direct and indirect N_2O emissions from inorganic nitrogen applied to soils in proportion to total inorganic nitrogen use.

Emissions from sewage sludge applied to soils were scaled in proportion to human population growth, following Lamb et al.,[25] and were allocated to each country in proportion to 2015 population size. Finally, emissions from crop residues were scaled (and allocated to each country) in proportion to changes in the total tonnage of crop production, following Lamb et al.,[25] assuming that the fraction of crop residue retained after harvest remains unchanged in the future.

Other agricultural emissions

Baseline CO_2 emissions from liming were taken from Brown et al.[107] and scaled in proportion to the total tonnage of crop production, following Lamb et al.[25] Baseline CO_2 emissions from the breakdown of urea-based fertilizers were taken from the same source and scaled in proportion to total inorganic N use (calculated as above).

We derived baseline CO_2, N_2O, and CH_4 emissions from farm energy use from Brown et al.,[107] corresponding to emissions from stationary and mobile combustion in agriculture and forestry. We scaled emissions from farm energy use in proportion to changes in the total tonnage of crop and livestock production.

The remaining emissions sources are attributed to other sectors under official GHG accounting protocols, but we include them here for completeness.

We assumed additional emissions from machinery manufacture and maintenance equivalent to 35% of farm energy use, following Lamb et al.[25]

N_2O and CO_2 emissions from the manufacture of synthetic nitrogen were taken from Williams et al.[137] for four forms of nitrogen: ammonium nitrate, urea, calcium ammonium nitrate, and urea ammonium nitrate. The approximate fraction of synthetic nitrogen applied in each form was estimated from the British Survey of Fertiliser Practice,[138] and emissions were scaled in proportion to total inorganic N use (calculated as above).

Per-hectare emissions (reported as CO_2e) associated with pesticide manufacture and breakdown were taken from Lamb et al.,[25] and scaled in proportion to the total cropped area.

Finally, we estimated embodied emissions (from farming, land-use change, processing, and transporting) in imported feed assuming 1.54 t CO_2e per MJ of feed.[25]

Peatland emissions

Following Evans et al.,[106] we used a combination of tier 1 and tier 2 emissions factors to estimate fluxes of CO_2 (including from dissolved and particulate organic carbon), CH_4 (including from ditches), and N_2O for each of 16 peatland condition categories. These condition categories represent woodland, cropland, intensive and extensive grassland, industrial and domestic peat extraction, rewetted and near-natural bog and fen, and six categories of modified bog.

Next, we paired each land cover category to one or more associated peatland condition categories (Table S15), and calculated the proportion of each land cover category attributable to each peatland condition category in each country, according to the 2013 areas presented in Evans et al.[106] We then calculated the weighted average GHG flux for each land cover category, according to the proportion of each land cover category attributable to each peatland condition category.

For our baseline scenario (2015), total estimated emissions from peatlands summed to 21,629 kt CO_2e, which is within 7.0% of the equivalent estimate in Evans et al.[106] Under future scenarios, we assumed that restored peat (either fen or bog) immediately transitions to rewetted fen or rewetted bog (although in practice the transition may take time, and fluxes should eventually approach those of near-natural peatlands). Future scenarios also imply a shift in the condition of existing bog, with extraction sites and eroded bog being restored to rewetted bog.

For paludiculture and continued arable production with water table management, we estimated direct CO_2 and CH_4 emissions using Equations 2 and 3 in Evans et al.,[139] assuming a mean water table depth of 10 cm for paludiculture and 40 cm for water table management. CH_4 from ditches and CO_2 from dissolved and particulate organic carbon were assumed to be the same as for extensive grassland for paludiculture and cropland for water table management. For both paludiculture and water table management, N_2O emissions were taken as the mid-point between improved grassland and cropland.

Emissions from saltmarsh and new intertidal habitat

According to our modified land cover map, the total baseline extent of saltmarsh in the UK is 71,373 ha. This is 60% greater than the 44,512 ha reported by Office for National Statistics.[140] Because of this discrepancy, we based our estimate of baseline (2015) GHG flux from saltmarsh on the officially recognized 44,512 ha, from which we subtracted the estimated 105 ha of saltmarsh projected to be lost each year due to sea-level rise and coastal squeeze.[101] We assumed an annually constant flux of -2.63 t CO_2/ha/year.[141]

For each hectare of newly created inter-tidal habitat, we assumed 0.79 ha of saltmarsh and 0.21 ha of mudflat based on the resulting ratio at existing managed realignment sites (M. Ausden, personal communication). For the saltmarsh fraction, we assumed the same annual net flux as for existing saltmarsh. For the mudflat fraction we assumed an annually constant net flux of -0.59 t CO_2/ha/year.[140] GHG fluxes from other intertidal habitats (including existing mudflat) were ignored.

Woodland carbon

We modeled carbon dynamics of new and existing woodland (including SRF), agroforestry, and wood pasture as described above for biomass production. We first converted annual estimates of standing biomass and debris into annual age-specific increments (i.e., negative fluxes representing sequestration). For biomass removed through either periodic thinning or rotational clear-fell, as well as for debris remaining after clear-felling or deforestation, we calculated decay profiles (i.e., positive fluxes) following Morison et al.[132] Tables S12, S16, and S17 give additional information on biomass fractions and decay parameters for harvested wood products and debris for Sitka spruce.

In Note S1 we explore different assumptions of harvested wood product longevity. We ignored potential substitution effects arising from the use of biomass in place of more carbon-intensive materials for energy or construction, as our focus here is the land sector and substitution effects are hard to quantify or guarantee (but see Forster and co-workers[38,70,142]).

Carbon in hedges

We assume that biomass in existing hedgerows is at equilibrium, with removals due to management canceling out any new growth. For new hedgerows, we assume that total biomass stock reaches 34.86 t C/ha,[107] or 5.23 t C/km (assuming 1.5 m width) after 20 years. This implies an annual flux of -0.26 t C/km for the first 20 years following hedge creation, and thereafter reaches equilibrium.

For hedge creation on arable land (representing 50% of all new hedgerows), we assumed an increase in soil carbon stocks, applying the same method as below for land-use change assuming a country-specific soil carbon density for hedgerows equivalent to the mid-point between grassland and woodland.

GHG emissions from perennial bioenergy crops

For *Miscanthus* and SRC (both restricted to cropland), we estimated carbon sequestration in soils and biomass, as well as emissions from fertilizer use. We assumed long-term average biomass stocks of 5.5 t C/ha for *Miscanthus* and 8.76 t C/ha for SRC, which we assumed were reached after 3 years.[71] Annual changes in soil carbon were derived from Richards et al.,[143] who report total soil carbon flux over 35 years following conversion of rotational cropland to *Miscanthus* (-55.4 t CO_2/ha) and SRC (-18.7 t CO_2/ha). We annualized these carbon fluxes by dividing by 35, assuming no additional change after 35 years. Finally, we took typical fertilizer application rates from Richards et al.[143] (30 kg N/ha/year for *Miscanthus*, 60 kg N/ha/year for SRC, and 45 kg N/ha/year for SRF), and assuming no fertilizer use for the first 2 years of each 20-year rotation for *Miscanthus* and SRC, or 26-year rotation for SRF. We treated GHG emissions from synthetic nitrogen as above for agriculture.

Following Thompson et al.[71] we assumed no Carbon Capture and Storage following combustion of biomass and did not account for avoided GHG emissions through the potential displacement of fossil fuels by bioenergy.

Soil and biomass stock changes following land-use change

For land-use transitions resulting from urban expansion, woodland creation, and semi-natural grassland creation, we estimated changes in soil carbon stocks according to country-specific weighted average changes in equilibrium soil carbon density following transition between forested land (representing broadleaved and coniferous woodland), cropland (representing arable and horticulture), grassland (representing improved grassland and all other semi-natural habitats), and settlement (representing built land).[107] We estimated the change in soil carbon stock following land-use change using the equation:

$$C_t = Ceq_{i,j} \times 1 - e^{t/(k/\ln(0.01)}$$

where C_t is change in carbon stock (t C/ha) in year t since the land-use transition, Ceq_{ij} is the country-specific change in equilibrium carbon stock (t C/ha) following transition between land-use i and j, and k is the time taken (in years) to reach 99% of the equilibrium carbon stock. Following Brown et al.[107] we assume that soil carbon gain is slower than soil carbon loss, setting k to 100 years for losses, 200 years for gains in England, Wales, and Northern Ireland, and 525 years for gains in Scotland. We also assumed that silvoarable agroforestry increased soil carbon under rows of trees, with the final equilibrium carbon stock equivalent to the mid-point between grassland and woodland (as for hedgerows on arable land).

We estimated equivalent changes in non-forest biomass as a result of urban expansion, woodland creation, and semi-natural grassland creation, assuming an instant change in biomass stocks following land-use change. Equilibrium biomass carbon density for annual cropland (representing arable and horticulture), shrubby grassland (representing dwarf-shrub heath), and non-shrubby grassland (representing improved grassland and all other semi-natural habitats) were taken from Brown et al.,[107] and we assumed an instant change in biomass stocks following land-use change.

Estimating breeding bird habitat availability

Our bird habitat evaluation models estimate breeding habitat availability for 109 native terrestrial birds under each scenario as a function of changes in land cover.

Bird-habitat models

The UK Breeding Bird Survey is coordinated by the British Trust for Ornithology and involves twice-yearly visits by experienced volunteers to 1-km survey squares during the breeding season. Squares are typically surveyed by walking a 2-km transect, divided into 10 sections. Adult birds are recorded within three distance bands from the transect line: 0–25, 25–100, and >100 m. We excluded records of birds from the final distance band, as well as fly-overs (for which calculating effective survey area, below, is impossible; following Lamb et al.[23]). In total, our dataset included records from 5,030 individual survey squares between 2013 and 2017.

Next, we used distance methods[144] to estimate habitat- and visit-specific detection probabilities for each species. We fitted Half Normal distributions to counts from the first two (bounded) distance bands, including visit (early or late) and habitat (assigned by surveyors to each 200-m transect section) as covariates.[145] We assumed that detection probability = 1 when distance = 0. Next, we calculated the weighted-mean detection probability for each species and each square, by averaging across early and late visits and across habitat types in proportion to their representation within each square.

We calculated effective survey area for each square and each species as the product of number of visits (max = 10 if a square was visited twice in each year 2013–2017), number of transect sections (max = 10), 0.04 (the surveyed area of each transect section, in km^2), and species- and square-specific detection probability (calculated as above, range = 0–1). Effective survey area was used an offset in the habitat-abundance models below.

We then constructed species-specific land cover-abundance models to predict the density of each species as a function land cover. We use an ensemble of two distinct modeling approaches.

"Method A" used a General Additive Model using the R package mgcv v.1.8-35,[146] assuming a negative binomial error structure to account for over-dispersion. The dependent variable was the count of species s across all visits to square x, and we used (log) effective survey area as an offset (such that predicted values correspond to effort- and detection-corrected densities). Independent variables were landscape-scale cover of arable and horticulture, woodland (coniferous and broadleaved combined), and built land (extracted from circles with a 2-km radius buffer around the centroid of each square),

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and local-scale cover of arable and horticulture, improved grassland, broad-leaved woodland, coniferous woodland, semi-natural grassland (acid, neutral and calcareous combined), bog, heather, fen, built land, freshwater, and coastal (extracted from each 1-km square). We also included total hedge length (within each 1-km square, excluding five squares in Northern Ireland with missing data). We assumed no interaction effects between covariates. To model residual spatial variation in abundance, we fitted a thin plate regression spline on latitude and longitude, fixing the maximum degrees of freedom to 20.

"Method B" uses an optimization-based mixture model approach to estimate the density of each species in each land cover category, accounting for the additive effects of NUTS1 region, landscape-level land cover, and hedge density. The basic form of the model is as follows:

$$\frac{count_{i,j}}{EA_{i,j}} = \sum p_{x,j} \times exp(d_{i,x})$$

where *count* is the total count of species *i* in each square *j*, *EA* is the effective area (accounting for survey effort and detectability), *p* is the proportion of each land cover *x* in each square, and *d* is the unknown density of each species in each land cover. We estimated the value of *d* using optimization via the R package Rcgmin, minimizing the quasi-likelihood (*qL*):

$$y = \sum p_{x,y} \times \exp(d_{i,x})$$

$$qL = -\sum \frac{count_{i,j}}{EA_{i,j}} \times \log(y) - y$$

For both models, we considered four alternative model structures, and selected for each species the model with the lowest AIC (method A) or *qL* (method B) value. The most complex model structure (1) included all covariates, and we tested simpler versions by removing (2) the hedge length covariate, (3) all three landscape-scale land cover covariates, or (4) the hedge length covariate and all three landscape-scale land cover covariates.

Apart from method B including coarser spatial information (regional effects rather than a 3D spline), the fundamental difference between the two approaches is that the parameter estimates for method B represent densities of each species in each land cover, whereas the parameter estimates for method A represent the marginal increase in abundance when a given land cover increases in area.

After excluding seabirds, non-native species and those with breeding populations of <50 pairs,[147] we initially considered 161 breeding bird species (Table S18). We identified the conservation status of each species following Eaton et al.,[148] and identified species occurring on woodland and farmland bird indicator lists.[149] Of these 161 species, detection models could not be fit for 12, and a further 24 were recorded at 25 or fewer survey squares so were excluded. We attempted to fit habitat-abundance models for the remaining 125 species; method A models failed to converge for 16 of these, leaving 109 focal species with successfully fitted habitat-abundance models (Table S18). Among species for which we could not fit method A models, 4 appear on the woodland specialist list (leaving 33 out of 37) and 1 on the farmland specialist list (leaving 18 out of 19).[149] Method B was successful for all 125 species, but our main results consider only the subset of 109 species for which predictions were available for both methods.

Scenario evaluation

To predict changes in population size under each scenario, we first extracted the covariates described above for the whole of the UK under each scenario (for years 2015 and 2050). We created a uniform 1-km grid to represent local-scale variables and buffered this by 2.5 km to represent landscape-scale variables. In the absence of alternative data, we treated paludiculture, *Miscanthus*, and SRC as arable, and horticulture and SRF as coniferous woodland. For Northern Irish squares with missing hedge data, we used the average hedge length from all other Northern Irish squares. We then combined these scenario-specific data with species-specific habitat-abundance models to predict species density in each 1-km square, which we summed to estimate total potential population size. To remove some unrealistically high predicted population densities from method A, we clipped the predicted density of each 1-km square so that no value exceeded the 99th percentile of predicted values. Finally, we expressed the 2050 predicted population size as a ratio of the 2015 predicted population size, and calculated the geometric mean across all species, red- and amber-listed species,[150] and farmland and woodland specialists and generalists.[149] For species with no breeding evidence in Ireland (Table S18; Balmer et al.[99]), we assumed populations of 0 in Northern Ireland under all scenarios.

To predict the population consequences of hedge creation, we multiplied hedge length by 1 (*Low* ambition), 1.4 (*Medium* ambition), or 1.5 (*High* ambition, representing a 50% increase in hedge length by 2050) when making predictions. We treated agroforestry and wood pasture as high-density hedges, by adding to each 1-km square 0.333 km of hedge (equivalent to 33-m spaced rows) per ha of new agroforestry or wood pasture.

To predict the population consequences of organic farming, we derived from the literature[151–155] estimates of breeding abundance in organic farmland relative to conventional farmland for 23 species (2 farmland generalists, 12 farmland specialists, and 9 woodland generalists). For these 23 species only, we took the predicted species-specific relative abundance in each square and inflated (or deflated) by the species-specific organic multiplier, multiplied by the fraction of each square under organic land. Note that method A does not allow us to derive species abundances for arable land cover (rather the increase in abundance expected per unit increase in arable cover within a 1-km square), but the approach described here produces sensible results.

Finally, for all species making up the farmland indicator list (including both specialists and generalists),[149] we modeled the (log-transformed) UK population index (or English index where this provided a longer time series)[156] as a function of UK average wheat yield from Defra.[113] For species where the effect of wheat yield was significant (ANOVA, $p < 0.05$), we predicted the species-specific relative population index under increased yields. We used the product of species-specific relative abundance and square-specific fraction of (non-organic) arable and horticulture and improved grassland as multipliers to inflate (or deflate) estimated abundance in each square.

SUPPLEMENTAL INFORMATION

Supplemental information can be found online at https://doi.org/10.1016/j.oneear.2023.09.005.

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AUTHOR CONTRIBUTIONS

Conceptualization, T.F., R.B.B., T.B.-L., G.M.B., W.J.P., and R.H.F.; methodology, T.F., T.B.-L., J.P.C., D.M., P.S., and R.H.F.; software, T.F.; formal analysis, T.F.; resources, D.M.; data curation, T.F.; writing – original draft, T.F.; writing – review & editing, R.B.B., T.B.-L., G.M.B., J.P.C., D.M., P.S., W.J.P., and R.H.F.; visualization, T.F.; supervision, W.J.P.

DECLARATION OF INTERESTS

The authors declare no competing interests.

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