The greenhouse gas impacts of converting food production in England and Wales to organic methods

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Agriculture is a major contributor to global greenhouse gas (GHG) emissions and must feature in efforts to reduce emissions. Organic farming might contribute to this through decreased use of farm inputs and increased soil carbon sequestration, but it might also exacerbate emissions through greater food production elsewhere to make up for lower organic yields. To date there has been no rigorous assessment of this potential at national scales. Here we assess the consequences for net GHG emissions of a 100% shift to organic food production in England and Wales using life-cycle assessment. We predict major shortfalls in production of most agricultural products against a conventional baseline. Direct GHG emissions are reduced with organic farming, but when increased overseas land use to compensate for shortfalls in domestic supply are factored in, net emissions are greater. Enhanced soil carbon sequestration could offset only a small part of the higher overseas emissions.

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Organic production

 Restraints and available N, second, the GHG impact of overseas land production imposed by the supply of livestock feed, rotational constraints and available N; second, the GHG impact of overseas land use associated with increased food-imports, and third, the GHG offset potential of soil carbon (C) sequestration under organic production. We also estimate uncertainties in our calculations using Monte Carlo analyses. In doing so we provide the most comprehensive national-scale assessment to-date of the potential land use, production and GHG impacts of up-scaling organic agriculture.

Results

Predicted food production. We predict a drop in total food production expressed as metabolisable energy (ME) by of the order of 40% compared to the conventional farming baseline (Fig. 1, Supplementary Table 1). Human edible protein outputs decreases by a similar proportion (Supplementary Table 2). The decrease is due to smaller crop yields per unit of land area under organic management, and the need to introduce fertility-building grass leys with nitrogen-fixing legumes within crop rotations. The latter requirement is a farming system-level effect that is not captured in crop-level comparisons16–18.

Figure 1 also shows large shifts in the combination of crops grown and numbers of animals reared. Increased diversity of crop

a Crops

b Livestock

Fig. 1 Projected food production under conventional and organic farming methods. a Crop production and areas. *oilseed rape. b Livestock production and numbers. **sheep numbers × 10, ***poultry numbers × 100, ****milk production in Mt × 10^5. Conversion to 100% organic methods caused decreases in wheat, barley, oilseed rape, pigs, eggs, poultry meat and milk, and an overall decrease to 64% of the conventional baseline. Data of Smith et al.16. Source Data are provided as a Source Data file.
rotations under organic management means total vegetable production is maintained. Edible protein production increases in arable areas, particularly in the east and north east of England, through increases in ruminant livestock and legume production. Production of organic oilseed rape (OSR) decreases substantially, primarily because of a much smaller cultivated area due to the relatively low yield of organic OSR compared to both conventional OSR and organic alternatives. The increase in legume and potato production is a result of an increase in the cultivated area: legumes for biological N fixation and potatoes both for weed control and because of their high ME yield. The area would have increased further had the constraint on maximum production in the model not been reached, which we set at 150% of current supply to reflect limits on consumer demand. Total sugar beet production decreased, but, due to its high ME yield, it reached its upper local limit in parts of eastern England, which we imposed to restrict expansion away from major processing centres. For most crops, the projected decreases in output are considerably greater than might be expected solely from the displacement of crops with ley in organic rotations. The production of minor cereals, such as oats and rye, increases, but this is not sufficient to offset the losses of wheat and barley.

Numbers of grazing livestock (sheep and beef cattle less dairy) increase, because of the increase in feed availability from legumes. But the volume of meat produced did not increase in proportion, as a result of lower carcass weights and longer finishing times under organic management. Numbers of monogastric livestock (pigs and poultry) and associated meat production fell sharply as a result of lower stocking rates and availability of concentrated feed. Dairy cattle numbers and milk production decrease due to greater reliance on concentrated feeds than grazing livestock and hence greater sensitivity to N availability, cropping area and cereal yields.

**GHG emissions per unit production.** Figure 2a shows estimated GHG emissions per unit of production for individual crops. The lower GHG emissions under organic cropping are largely due to replacement of N fertiliser with biological N fixation in legys, resulting in less CO₂ and N₂O from fertiliser manufacture and less N₂O per unit of production. We concentrate on N in our analysis, and not on other plant nutrients, because N is required in the greatest quantities and its inputs and outputs are the most sensitive to differences between conventional and organic systems. However, balances of P, K and other nutrients must also be maintained, and we therefore account for the GHGs associated with extracting and applying the P and K minerals commonly used in organic systems to maintain balances.

Emissions per unit production are greater for some organic crops, such as field beans, due to increased N leaching and nitrification-denitrification losses, because more must be grown on heavy wet soils. However, a large proportion of field beans grown would have to be exported because of low rates of domestic consumption, and we allow for this in the model with a maximum limit on production, as for potatoes. Oats and spring barley, which require less manufactured N fertiliser than other cereals, have greater GHG emissions per unit production under organic management because yields are smaller. Lower marketable yields in organic potato cropping also lead to greater emissions per unit of product. Emissions are also greater for organic crops requiring higher fossil fuel input in their cultivation, such as organic carrots requiring flame weeding.

Figure 2b shows emissions per unit of production for individual livestock types. Organic pig production results in lower GHG emissions per unit of production because outdoor organic systems use less fossil energy in housing and there are no CH₄ emissions from slurry storage; however, N₂O emissions increase as a result of greater leaching and denitrification from organic manures. In common with previous studies, we find that poultry meat and egg production generates greater emissions under organic management due to poorer feed conversion ratios, longer rearing times, higher mortality rates and greater leaching losses compared to conventional free range and fully housed systems. Organic dairy, beef and sheep production results in lower total GHG emissions per unit of production, as a result of the increased efficiency of forage production under organic management, although greater forage intake increases the total CH₄ contribution.

**National GHG emissions.** Figure 3 gives the aggregated national emissions. It shows that the direct emissions associated with organic crop (Fig. 3a) and livestock (Fig. 3c) production are smaller for organic farming compared with conventional: by 20% for crops, 4% for livestock and 6% overall. This is a slightly lower estimate of the effect of conversion to organic farming than in Audsley et al.’s study. The decrease occurs despite an increase in transport emissions, illustrating the relatively small contribution that transport makes to agriculture’s total GHG budget.

However, the picture is very different when we allow for, first, CO₂ emissions from land use change overseas to make up for shortfalls in home production under organic methods, and second, enhanced soil C sequestration under organic methods at home and overseas, as shown in Fig. 3b, and 3d for different ways of making these allowances. The next two sections give our rationale for how we have done this.

**Soil carbon sequestration.** Carbon sequestration rates are expected to be greater under organic farming because of greater use of manures and slurry linked to more integrated management of livestock and crops, and longer crop rotations with legumes involving forage legumes. Although in conventional systems there is generally a greater separation of livestock from crops, farmyard manures will mostly be applied to land somewhere, so the net transfer of C from the atmosphere to land would be about the same. On the other hand, excessive manure applications in livestock-dense areas under conventional management lead to over-fertilisation and suboptimal C sequestration. Although we found livestock production decreased under organic management, total livestock numbers were not much different and there was a substantial shift to grazing animals with 61% more sheep and 14% more cattle (beef plus dairy; Fig. 1). We estimate there would be approximately 12% more farmyard manure as a result (Supplementary Table 3).

We estimate potential C sequestration under organic management using rates of change in soil C derived from the National Soil Inventory of England and Wales for different land use classes by Kirk and Bellamy, and assuming the change from conventional to organic farming was equivalent to a change from continuous arable cropping to rotational grass (Methods). This gives sequestration rates of 0.28 Mg C ha⁻¹ yr⁻¹ for arable land converted to rotational grass, or, after adjusting for the proportion of arable to arable plus rotational grass across England and Wales, 0.18 Mg C ha⁻¹ yr⁻¹. We used this as the upper rate in the calculations for Fig. 3. For comparison, in a literature review of experiments comparing conventional and organic farming, Gattinger et al. found sequestration rates between 0.07 and 0.45 Mg C ha⁻¹ yr⁻¹. However, most of these comparisons involved very high rates of external organic matter inputs to the organic systems, up to 4 times those under conventional farming. Given that we found only 12% more farmyard manure...
under organic farming, Gattinger et al.’s higher estimates are unrealistic. We therefore use Gattinger et al.’s lower value as the moderate rate in Fig. 3. It should be noted that the bulk of any C sequestration will be limited to the first decade or two following conversion, because any given soil has a finite capacity to accumulate C depending on its characteristics and local environmental conditions. A new steady-state soil C content will be reached after a few decades when rates of decomposition in the soil at the higher C content match the increased rates of C inputs.

**Overseas land conversion.** We estimate that the land area needed to make up for shortfalls in domestic production is nearly five times the current overseas land area used for food for England and Wales (Fig. 4). Total agricultural land-use is therefore 1.5 times greater than the conventional baseline (combining domestic and overseas land). This is considerably greater than the 16–33% increase in land requirements projected in a recent study of global conversion to organic farming. The difference reflects the high conventional crop yields and livestock productivity in the UK compared with countries using less intensive, lower-yielding...
farming, and the correspondingly greater production penalties in conversion to organic methods.

The consequences for net GHG emissions will depend on the nature of the land use change. If it entails conversion of existing natural or semi-natural vegetation or pasture to crops, the cost will be greater than for increased production from existing arable land, which will have already lost C compared with its original natural state, and which might be expected to sequester some C from the atmosphere under organic management. The emissions associated with land use changes will apply over a similar period to the potential gains from enhanced soil C sequestration (i.e., a few decades). We compare three ways of assessing this and associated soil C sequestration: first, if all the additional production is on land formerly under grass, with no associated C sequestration; second, if half the additional production is on land formerly under grass, with a low rate of C sequestration; and third, if a quarter of the additional production is on land formerly under grass, with a high rate of C sequestration (Methods).

In addition, there is the opportunity cost of the amount of C that could be sequestered if the land were instead used to...
maximise its C storage potential, for example by converting it to productive forest. This aspect is considered by Searchinger et al.35, who define a ‘Carbon Opportunity Cost’ (COC) as the amount of C that could be sequestered annually per kg of agricultural commodity if the land were instead used to regenerate forest. We also calculated this (Methods).

The results (Fig. 3b, d and Table 1) show that the net effects are sensitive to both the LUC scenario and the degree of soil C sequestration. If all the LUC is by conversion of grassland with no C sequestration (the High scenario), net emissions increase by 56% over the conventional baseline. Whereas, if only 25% of the LUC is from grassland, with a high rate of C sequestration (the Low scenario), net emissions are comparable to those in the conventional baseline. With 50% LUC from grassland, and a moderate rate of C sequestration (the Medium scenario), the net increase is 21%. However, if the COC is added in, the net GHG costs of organic production are much worse. For the Medium LUC and C sequestration scenario, adding in the COC (35.7 ± 6.6 Mt CO₂ yr⁻¹) gives a net increase in emissions over the conventional baseline of 1.7 times.

**Discussion**

The results show that widespread adoption of organic farming practices would lead to net increases in GHG emissions as a result of lower crop and livestock yields and hence the need for additional production and associated land use changes overseas. It is not obvious how additional overseas land could be found, without expanding the existing area of tilled land by ploughing up grassland. The global demand for food is expected to increase by 59–98% by 205034. Given that land resources are finite, this implies more competition for land, and more-intensive food production per unit land area, whereas current organic systems are inherently less intensive.

There are undoubted local environmental benefits to organic farming practices, including soil C storage, reduced exposure to pesticides and improved biodiversity. However, these potential benefits need to be set against the requirement for greater production elsewhere. As well as increased GHG emissions from compensatory changes in land use to make up for production shortfalls, there are substantial opportunity costs from reduced availability of land for other purposes, such as greater C storage under natural vegetation35. Further, although organic systems may favour increased local biodiversity, habitat fragmentation under low-yielding organic systems may mean global species diversity is in fact greater under land-sparing, high-yielding systems36,37.

Could yields under organic management be improved to reduce land requirements? Improvements in organic rotation design and more effective and reliable supplies of N from biological fixation are possibilities38,39. However, these improvements are probably marginal, given the fundamental requirement for more leys in rotations under organic management. Given the much larger contribution of livestock farming to GHG emissions, a greater impact could be gained from reduced meat consumption. Less livestock farming could release land for crops for human consumption and for other purposes such as C storage40. However, against this, global trends are towards greater per capita and total meat consumption33. Also livestock can play important roles in local nutrient cycling and the provision of ecosystem services41,42.

In summary, our assessment of the impacts of a 100% conversion to organic farming in England and Wales has revealed that, whilst improvements in resource use efficiency could be obtained, reduced outputs would mean that more imports would be required to maintain food supplies. This major expansion in agricultural cultivation overseas to make up for domestic supply shortfalls would lead to increased GHG emissions from the associated land use changes. Ultimately it is unlikely that there exists any single optimal approach to achieving environmentally sustainable food production. Therefore, context-specific evaluations are required to reveal the extent to which organic systems can contribute, alongside other approaches, to multi-objective and internationally binding sustainability targets.

**Methods**

The OLUM. The OLUM (Optimal Land Use Model)16 is a linear programming (LP) model that includes a suite of organic farming activities that take place in nine Robust Farm Types: specialist cropping, mixed arable and livestock, specialist

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**Table 1 Total GHG emissions from crop and livestock production under conventional and organic production allowing for High, Medium and Low levels of overseas LUC and soil C sequestration as in Fig. 3**

|                     | Conventional | Organic |
|---------------------|--------------|---------|
|                     | High         | Medium  | Low     |
| Emissions (Mt CO₂ yr⁻¹) | 49.3 ± 2.1   | 77.1 ± 4.2 | 59.8 ± 2.7 | 46.6 ± 4.1 |
| Fraction as CO₂ (%)   | 34           | 59      | 48      | 33       |
| Fraction as CH₄ (%)   | 36           | 25      | 32      | 41       |
| Fraction as N₂O (%)   | 29           | 16      | 21      | 26       |
| Difference from conventional baseline | *p < 0.05* | NS      | NS      |

*Data are means ± 1 std. dev*
dairy, lowland grazing livestock, Less Favoured Area (LFA) grazing livestock, pigs and poultry, and other. These cover the entire agricultural land-base in England and Wales. The proportion of the function of these crops subject to constraints on resource availabilities, is the sum of total crop and livestock production, expressed as ME. Although human diets also need proteins, fats and nutrients, energy requirements are deemed to be a primary driver of consumption and an inadequate food-energy intake is almost always accompanied by insufficient intake of nutrients. The basic formulation of the OLUM is

\[
Z = \sum_{i=1}^{n} \sum_{j=1}^{m} C_{ij} x_{ij} \text{subject to } Rx_{ij} \leq b, x_{ij} \geq 0, \tag{1}
\]

where \( Z \) is the objective function to be maximised, \( C_{ij} \) is the ME output (fresh weight per unit crop area or livestock number yr\(^{-1} \)) of agricultural product \( i \) on soil \( x \) in season \( j \), \( x_{ij} \) is a scalar for the agricultural activity (crop area or livestock number), \( Rx_{ij} \) is a factor for the input and resource requirement associated with the agricultural activity, and \( b \) is a vector for resource endowment and resource availability (e.g., land by soil and rainfall class, and available soil N). Human dietary change is not considered.

In each farm type, the set of crop and livestock production activities available are fixed, as evidence suggests that the dominant agricultural activity (e.g., dairy farming) will usually stay in place post conversion to organic management, due to existing farm infrastructure, farming knowledge and local conditions. However, these activities can be individually expanded and contracted endogenously. The land areas under each farm type are fixed, reflecting the area coverage of their conventional equivalents recorded in the June Survey of Agriculture (see data sources for export volumes in Supplementary Table 5). Where the OLUM derives crop areas, by each soil and rainfall class, from the National Soil Inventory (www.LandIS.org.uk). Four rainfall classes are defined based on 30-year Meteorological Office annual rainfall data: dry 539–635 mm, medium 636–723 mm, wet 724–823 mm and very wet 824–2500 mm. The total areas of each soil \( x \) rainfall combination were determined by identifying the dominant combination in each 5 km \( \times \) 5 km grid square and allocating to that combination the sum of the areas of each square, less any non-agricultural area. The OLUM produces a best estimate of production under fully organic agriculture in England and Wales, assuming that food production would be maximised. To ensure that the results are reasonable, outputs are compared to the real-world distribution of conventional production in 2010 derived from a range of each 5 km \( \times \) 5 km grid square and allocating to that combination the sum of the crops and livestock products by GHG coefficients derived from Hess et al. Transport burdens for imported sugar and sheep meat are derived from Plassman et al. and Webb et al., respectively.

Where the OLUM generates crop and livestock production in excess of domestic demand, the surpluses are assumed to be exported and the GHG and fossil energy burdens associated with production of the exported commodities are subtracted from the total environmental burdens of organic agriculture. The same adjustment is made to the GHG estimates of exports for conventional agriculture (see data sources for export volumes in Supplementary Table 5). Where the OLUM reduces production below the level of domestic demand it is assumed that no exports occur, i.e., domestic consumption would take priority.

Fossil energy use and GHG emissions associated with the production of oilseed rape, sugar beet, wheat and lamb from non-European countries are derived from Pelletier et al., Tailvikas et al. and Webb et al. The environmental burdens associated with crop and livestock products sourced from Scotland, Northern Ireland and the rest of Europe are derived from the Agri-LCA, under the assumption that similar emissions and fossil energy use would occur in these systems.

**Soil carbon sequestration.** We obtain an upper estimate of potential sequestration rates in organic systems based on rates of change of soil C measured in the National Soil Inventory (NSI). We use data from the Agri-LCA models to better reflect soil carbon change. Bel-lamy summarised the NSI results by fitting to the data the simple single-pool model

\[
dC/dt = -kC, \tag{2}
\]

where \( C \) is the C content per unit land surface area, \( I \) is the rate of input from vegetation and other sources and \( k \) is a rate constant for decomposition. They fitted Eq. (2) to the data for each NSI land use class separately, omitting organic soils (which accounted for <5% of all the soils in the NSI) because their rates of change were less certain. The soil C content at steady state, when \( dC/dt = 0 \), is equal to \( I/k \). Soils with C contents greater than the steady-state value lose C, those with C contents less than it sequester C.

We take the NSI class ‘rotational grass’ (i.e., grass that is sown and then tilled every few years as part of an arable rotation) to represent potential C contents under conventional arable management, and the class ‘arable crops’ under conventional arable management. The mean soil C contents were 43.2 (\( n = 552 \) sites) and 8.7 (\( n = 301 \) sites) Mg C ha\(^{-1} \) under arable and rotational
grass, respectively, and the calculated steady-state C contents were 37.6 and 35.0 Mg C ha\(^{-1}\), respectively, indicating the rotational grass soils were on average close to steady state. All of their C contents were therefore redefined, with mass transfers predicted from potential sequestration levels. The values of I and K for rotational grass were 2.54 Mg C ha\(^{-1}\) yr\(^{-1}\) and 0.046 yr\(^{-1}\), respectively (equivalent to negative emissions of \(-9.3\) and \(-0.17\) Mg CO\(_2\) ha\(^{-1}\) yr\(^{-1}\)). Substituting these values and the mean arable C content in Eq. (2) gives for the mean rate of sequestration on conversion from arable to rotational grass at (\(z = 0.07\) Mg C ha\(^{-1}\) yr\(^{-1}\)) after adjusting for the proportion of arable to arable plus rotational grass, the rate is \(0.28 \times 552/(552 + 301) = 0.18\) Mg C ha\(^{-1}\) yr\(^{-1}\) (or \(-1.03\) Mg CO\(_2\) ha\(^{-1}\) yr\(^{-1}\)). We use this as the high C sequestration rate in Fig. 3. We assume sequestration rates of established grass are zero and the potential sequestration rate rate to be zero given that these sites will have already reached steady state.

For comparison, in a literature survey of experiments comparing conventional and organic farming, Gattinger et al.\(^{24}\) found sequestration rates between 0.07 and 0.18 Mg C ha\(^{-1}\) yr\(^{-1}\). However, most of these comparisons involved very high rates of external organic matter inputs to the organic systems. The average inputs were four times the upper conventional farming for the full database and two times for systems with inputs equivalent to those from one European Livestock Unit (ELU) ha\(^{-1}\)\(\cdot\)yr\(^{-1}\). We calculate that quantities of farmyard manure would be only approximately 12% greater under organic farming, as a result of greater numbers of grazing livestock (Supplementary Table 3). We therefore consider Gattinger et al.’s upper and middle sequestration estimates to be unrepresentative and take as the moderate sequestration rate in Fig. 3 their lower value of 0.07 Mg C ha\(^{-1}\) yr\(^{-1}\).

Gains through C sequestration will be time-limited, because any given soil has a finite capacity to accumulate C and a new steady-state C content will be reached after a few years, when increased C inputs are matched by increased losses at the greater soil C content. Our estimated sequestration rates therefore only apply in the early-years following conversion to organic methods. Based on the NSI data, a new steady-state C content on conversion from arable to rotational grass would only be attained after (55.02 – 43.15)/0.28 = 42 years.

### Additional emissions from overseas LUC and C sequestration

We estimate the additional overseas land area required for each of the food products listed in Fig. 1, produced organically, as follows. For crops, we use first, regional yield data from Eurostat, second, organic crop yields from the recent meta-analysis by de Ponti et al.\(^{23}\) and third, results of an LCA for milling wheat grown in Canada\(^{38}\). For livestock we use first, regional yield data from Eurostat, second, results from the Agri-LCA\(^{28}\) and third, recent studies on the environmental burdens of imported lamb from New Zealand\(^{23,29}\). The additional land area is calculated from the total overseas area required less the amount required for imports in the conventional baseline (based on the values in Supplementary Table 6). The corresponding emissions are calculated as follows.

We assume that woodland would not be converted for food production as this would conflict with the principles of the International Federation of Organic Agriculture Movements (IFOAM)\(^{31}\). We calculate emissions from the conversion of grassland to crops from the area converted multiplied by LUC emission estimates specified by the British Standards Institute for a range of countries\(^{32}\). Considering that not all LUC would be from grassland, we compare three ways of assessing the net emissions from overseas LUC and associated soil C sequestration, plus that of home production, as follows. First, High: the additional land required is converted from grassland, with no net soil C sequestration at home or overseas. Second, Medium: 50% of the additional arable land is converted from grassland, with a moderate rate of C sequestration (0.07 Mg C ha\(^{-1}\) yr\(^{-1}\)) at home and overseas. Third, Low: 25% of the additional arable land is converted from grassland, with a high rate of C sequestration (0.18 Mg C ha\(^{-1}\) yr\(^{-1}\)) at home and overseas.

Following Searchinger et al.\(^{35}\), we also calculate the additional ‘carbon opportunity cost’ (COC) of using the land for agriculture as the quantity of C that could be sequestered annually if the average productive capacity of land used to produce 1 kg of each food product globally were instead devoted to regenerating forest. We calculate the total COC from Searchinger et al.\(^{35}\) COC factors per unit fresh weight of each food product (separating crops for human consumption from those used as animal feed) and the additional fresh weight of imports of each product required to offset home production shortfalls. This is in addition to the emissions calculated under the LUC and C sequestration scenarios (1)–(3) above. This ‘C gain’ method—as opposed to a ‘C loss’ method based on plant and soil C lost to date per unit food production—applies if it is only possible to increase C by re-establishing forests.

### Uncertainty analysis

Estimates of uncertainty for each main commodity analysed were produced following the method of Wiltshire et al.\(^{33}\). Uncertainties were derived using Monte Carlo simulations with each domestically produced crop commodity given an uncertainty estimate of 10% (i.e., in a triangular distribution with upper and lower bounds at 10% of the mean) and each domestically produced livestock commodity at 15%. The emissions for crops and livestock were summed in separate Monte Carlo simulations to produce overall uncertainty estimates for each sector (as the standard deviation). These were increased by 15% for all imported commodities in bloc. Emissions from import transportation were assumed to have a standard deviation of 10% of the mean\(^{33}\), i.e., the coefficient of variation (CV) is 10%. The areas of land derived by the LPI were assumed to have a mean increase of 1% which was varied randomly to provide the overall solution, not per crop, given that all areas were derived from any individual solution. Error bars on production area per crop (or livestock commodity) are thus not shown.

The final emissions and uncertainty estimates for each production system were derived from the sum of emissions from domestically produced crops and livestock together with emissions from imported crop and livestock production, together with their transport emissions, based on supply chain data from Webb et al. (2013)\(^{23}\) and Williams et al. (2017)\(^{24}\). Estimates of the uncertainty from LUC were derived from Houghton\(^{30}\) and those from C sequestration from Kirk and Bellamy\(^{30}\) for the upper rate and from Gattinger et al.\(^{24}\) for the medium and lower rates. These were implemented as the uncertainty being a proportion of the means that were applied to the LUC and C sequestration scenarios. These were established as having a CV of 17% for LUC\(^{35}\), which was increased for the carbon opportunity cost of Searchinger et al.\(^{35}\) by a factor of 1.5 to allow for the extra uncertainty of the method (i.e., CV of 26%). The uncertainty of the high level of C sequestration was 80 and 24% for lower levels.

The uncertainty estimates for the sum of crop and livestock commodities, transport and land use change emissions and sequestration are summarised in Supplementary Table 7. These were used as input values of uncertainties in the last stage to derive the overall uncertainties of each scenario. We tested the significance of differences in mean values, z, using Eq. (3),

$$z = \frac{|m_A - m_B|}{\sqrt{CV^2 x m_A^2 + CV^2 x m_B^2}} \times 100$$

where \(m_A\) and \(m_B\) are the means of systems A and B, respectively, and CV is the CV of each mean (expressed as whole numbers). The threshold for a significant difference at the 5% level was \(z = 1.96\).

### Reporting summary

Further information on research design is available in the Nature Research Reporting Summary linked to this article.

### Data availability

The data underlying these calculations can be accessed at: https://doi.org/10.6084/m9.figshare.6080333.v2. OLUM model code and data can be accessed at: https://tinyurl.com/yxdzxsrv. The Agri-LCA models and data can be accessed at: https://tinyurl.com/yy5jq7f7.

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