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RESEARCH ARTICLE

Large-scale deployment of grass in crop rotations as a multifunctional climate mitigation strategy

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Abstract

The agriculture sector can contribute to climate change mitigation by reducing its own greenhouse gas (GHG) emissions, sequestering carbon in vegetation and soils, and providing biomass to substitute for fossil fuels and other GHG-intensive products. The sector also needs to address water, soil, and biodiversity impacts caused by historic and current practices. Emerging EU policies create incentives for cultivation of perennial plants that provide biomass along with environmental benefits. One such option, common in northern Europe, is to include grass in rotations with annual crops to provide biomass while remediating soil organic carbon (SOC) losses and other environmental impacts. Here, we apply a spatially explicit model on >81,000 sub-watersheds in EU27+UK (Europe) to explore the effects of widespread deployment of such systems. Based on current accumulated SOC losses in individual sub-watersheds, the model identifies and quantifies suitable areas for increased grass cultivation and corresponding biomass- and protein supply, SOC sequestration, and reductions in nitrogen emissions to water as well as wind and water erosion. The model also provides information about possible flood mitigation. The results indicate a substantial climate mitigation potential, with combined annual GHG savings from soil-carbon sequestration and displacement of natural gas with biogas from grass-based biorefineries, equivalent to 13%–48% of current GHG emissions from agriculture in Europe. The environmental co-benefits are also notable, in some cases exceeding the estimated mitigation needs. Yield increases for annual crops in modified rotations mitigate the displacement effect of increasing grass cultivation. If the grass is used as feedstock in lieu of annual crops, the displacement effect can even be negative, that is, a reduced need for annual crop production elsewhere. Incentivizing widespread deployment will require supportive policy measures as well as new uses of grass biomass, for example, as feedstock for green biorefineries producing protein concentrate, biofuels, and other bio-based products.

KEYWORDS
agriculture, climate mitigation, environmental benefits, environmental impacts, Europe, grass, land use, perennial crops, soil carbon, spatial modelling

1 | INTRODUCTION

Global warming of 1.5°C (and 2°C) will be exceeded during the 21st century unless substantial reductions in greenhouse gas (GHG) emissions occur in the coming decades (IPCC, 2021). All global pathways that limit warming below 1.5 and 2°C (with no or limited overshoot) involve rapid or immediate GHG emission reductions using, for example, methods for carbon dioxide removal (CDR) from the atmosphere (IPCC, 2022; Minx et al., 2018). The agriculture sector, representing 22% of total net anthropogenic GHG emissions, can contribute to climate change mitigation by reducing GHG emissions and by CDR via carbon sequestration in vegetation and soils. The sector can also provide biomass for mitigation in the energy, industry, and transport sectors by substituting for fossil fuels and other GHG-intensive products (IPCC, 2019a, 2022). Meanwhile, the agriculture sector needs to address water, soil, and biodiversity impacts caused by historic and current practices (DeBoe, 2020; Tilman et al., 2002). The sector also needs to adapt to climate change, which is expected to cause new stresses on agricultural systems and exacerbate risks to human health, ecosystem health, food systems, and livelihoods (IPCC, 2019a).

The European Union (EU) Common Agricultural Policy (CAP) for the period 2021–2027 includes regulations and incentives to promote climate change mitigation, environmental protection, and preservation of biodiversity (European Commission, 2018a). Other EU policies that are likely to influence agricultural practices include the Renewable Energy Directive (European Commission, 2018b), the European Green Deal (European Commission, 2019), the Biodiversity Strategy for 2030 (European Commission, 2020a), and the Farm to Fork Strategy (European Commission, 2020b). Changing agricultural practices toward a greater share of perennial species, for example, perennial grasses and legumes (here, "grass"), in intensively cultivated agricultural landscapes can contribute to many of the objectives underlying these policies by providing biomass for food, bioenergy, and other bio-based products while reducing the environmental impacts from agriculture (Christen & Dalgaard, 2013; Englund et al., 2021; Englund, Börjesson, et al., 2020; Englund, Dimitriou, et al., 2020; Ferrarini et al., 2017; Styles et al., 2016).

Biomass production in species-rich mixtures of perennial grasses on marginal land has the potential to enhance biodiversity and carbon sequestration in soils (Carlsson et al., 2016). Another promising option is to include grass in crop rotations with annual crops in mixed farming systems, a common practice in cold or humid climates, primarily in northern Europe (Jarvis et al., 2017). This practice can have multiple environmental benefits, such as increasing the soil organic carbon (SOC) content, but can also enhance crop yields in the longer term (Prade et al., 2017). Interest is also growing in biorefineries that process grass–clover mixes into protein concentrate and a multitude of other products, for example, feed, fibers, heat, power, and biofuels (Njakou Djomo et al., 2020). For example, lactic acid bacteria can facilitate the use of grass biomass to produce a protein concentrate suitable for feeding monogastric animals as well as ruminants, with multiple co-products (Lübeck & Lübeck, 2019). Such solutions, using alternatives to high-input and high-emission annual grain and seed crops as feedstock, can enable sustainable intensification of the agricultural systems with reduced environmental impacts (Larsen et al., 2017).

Here, we estimate the effects of producing perennial grass in rotation with annual crops at large scale on biomass production, remediation of SOC losses from historic land use, and mitigation of additional environmental problems. We model the introduction of grass in crop rotations with annual crops in more than 81,000 sub-watersheds ("landscapes", see Section 2) across Europe (EU27 + UK). We then quantify grass biomass production—in terms of dry matter (DM), extractable protein, energy content, and biogas output—and increases in SOC and the corresponding GHG emission savings from carbon sequestration and fossil fuel substitution. Finally, we quantify or indicate multiple environmental co-benefits: (i) reduced wind erosion, (ii) reduced water erosion, (iii) reduced nitrogen emissions to water, and (iv) mitigated flooding events.

The results show that widespread deployment of perennial grass in rotation with annual crops would result in substantial carbon sequestration in agricultural soils. The annual carbon sequestration by 2050 in two illustrative deployment scenarios corresponds to about 5%–10% of current GHG emissions from agriculture in EU27 + UK. The combined annual GHG savings from soil carbon sequestration and biogas use are equivalent to 13%–48% of current GHG emissions from agriculture. Environmental co-benefits are notable—in some cases exceeding the estimated mitigation needs. There will obviously be notable changes in agricultural output where crop rotations are modified to include more grass and less annual crops. However, when considering yield increases for annual crops in modified rotations and the possibility to produce animal feed and other bio-based products from grass instead of annual crops, increased cultivation of grass in rotations dominated by annual crops can result in a negative displacement effect, that is, reduced need for annual crop production elsewhere.

2 | METHODS

We constructed a model to identify sub-watersheds ("landscapes") where the introduction of grass into crop rotations with annual crops could increase SOC. We designed three grass rotation options to include in the model and two scenarios (a high and a low estimate) for large-scale deployment using combinations of these rotations, with separate sets of conditions for the implementation of the two scenarios depending on the current accumulated SOC losses in the landscape. For each alternative and scenario, the model calculates the total area under grass production in each landscape and the corresponding grass biomass production, in terms of dry matter, energy (J), and protein (metric tons of extractable crude and true protein, respectively). Furthermore, the model estimates the corresponding SOC increases by 2030, 2050, and 2080, both relative to 2020 and relative to a business-as-usual scenario with a continuation of current land use. Finally, the model quantifies a number of co-benefits, that is, environmental benefits that do not incentivize implementation. These include (i) avoided soil loss by water, (ii) avoided wind erosion, (iii) avoided nitrogen emissions to water, and (iv) mitigated flooding events.

The analysis and aggregation unit is equivalent to sub-catchment or sub-watershed and is also referred to here as a "landscape." A previously published pan-European dataset containing >81,000 polygons (Englund, Börjesson, et al., 2020), based on functional elementary catchments from the ECRINS database (European Environment Agency, 2012), was used. For each landscape, we have previously estimated, for example, the area under annual crop production, degree of current environmental impact (nitrogen emissions to water, soil loss by water erosion and by wind erosion, recurring floods, and accumulated losses of SOC), and the estimated effectiveness of strategic perennialization in mitigating these impacts, in general terms (Englund, Börjesson, et al., 2020).

The term "landscape" is here defined as an intermediate integration level between the field and the physiographic region (Englund et al., 2017). The use of the term is considered appropriate since the anthropogenic processes (agricultural land use) within a sub-watershed, combined with hydrological processes that are constrained by a sub-watershed, determine (changes in) nutrient, water, and mass flows (Englund et al., 2021). Using the term landscape also clarifies that implementation and impact mitigation are enabled by measures taken by multiple stakeholders at a greater scale than the individual field, thus applying a "landscape perspective" (Dale et al., 2013).

All GIS operations, including all database aggregation queries, were done in GRASS GIS (GRASS Development Team, 2020) with projection EPSG:3035. All modeling, apart from input data preparation, was conducted using a Python script (Englund, 2022a) with a GUI that facilitates execution of selected modules. Complete input data are publicly available (Englund, 2022b). Cartography was done in QGIS (QGIS.org, 2020).

2.1 | Grass production in crop rotation systems and scenarios for widespread deployment

The model first selects landscapes where the effectiveness of strategic perennialization has been classified as "medium" or higher (Englund, Börjesson, et al., 2020), based on accumulated losses of SOC in combination with the density of annual crops (Figure 1). For each landscape, the model then makes calculations for three management alternatives where grass is included in crop rotations with annual crops and two scenarios for "widespread deployment." The management alternatives are:

- *2/6-grass*: Two years of grass added to a 4-year rotation of the most dominant crops in the region.
- *3/7-grass*: Three years of grass added to a 4-year rotation of the most dominant crops in the region.
- *4/8-grass*: Four years of grass added to a 4-year rotation of the most dominant crops in the region.

We constructed two scenarios for widespread deployment, where the introduction of grass in crop rotations is conditioned by the degree of accumulated SOC losses in each landscape:

- *Low estimate*: The 2/6-grass system is implemented on all fields currently under annual crop production where the accumulated SOC loss is classified as "very high," on 50% of all fields where it is "high," and on 25% of all fields where it is classified as "medium."
- *High estimate*: The 2/4-grass system is implemented on all fields currently under annual crop production where the impact is classified as "medium," the 3/7-grass system is implemented where it is "high," and the 4/8-grass system is implemented where it is classified as "very high."

2.2 | Grassland area and corresponding biomass and protein production

For the three production systems and two deployment scenarios, the average area under grass production in each landscape was calculated as the product of annual crop area (Englund, Börjesson, et al., 2020) and the share of grass relative to annual crops over time in the different systems, that is, 1/3 for the 2/6-grass system, 3/7 for the 3/7-grass system, and 4/8 for the 4/8-grass system. The

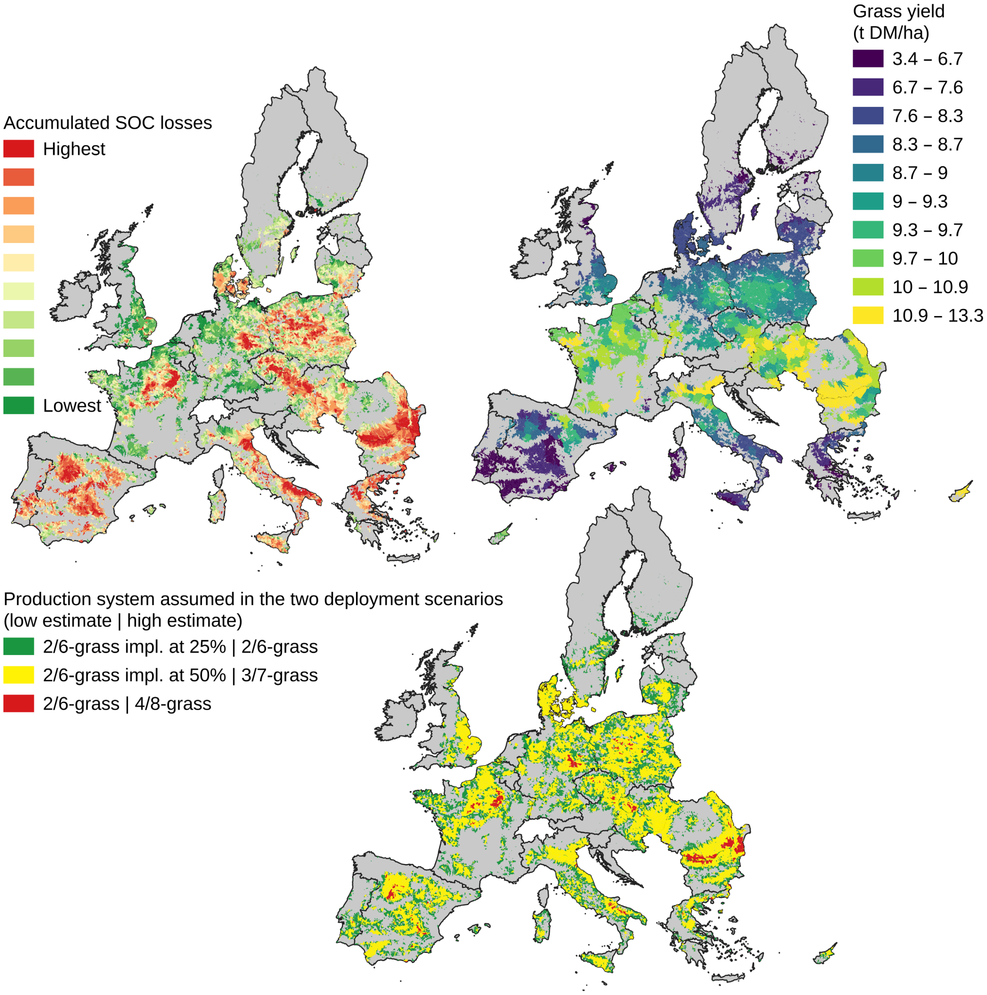

FIGURE 1 Accumulated soil organic carbon (SOC) losses (top left), simulated grass yields (top right), and production systems implemented in the two deployment scenarios (bottom), based on estimated effectiveness to remediate accumulated SOC losses.

area was then multiplied by 25% and 50%, respectively, to calculate the areas under 25% and 50% implementation, for the low-estimate scenario.

Having calculated the areas, the corresponding biomass production was estimated for each landscape by multiplying the area under grass production with simulated grass yields from a pan-European dataset at NUTS3 level (Dees et al., 2017). The average yield for miscanthus, switchgrass, and reed canary grass using a "medium" yield-input management level was calculated in each NUTS-3 region and identified for each landscape by first spatially joining landscapes to NUTS-3 regions, and then joining the database tables. The yields were then adjusted for each system assuming that the yield in the establishment year is 50% of subsequent yields (Moyo et al., 2016). Yields for the different systems were thus adjusted as follows. Yields are expressed as t DM ha year $^{-1}$ and are visualized in Figure 1.

- $2/6\text{-grass}_{yield} = (0.5 + 1)/2 \times \text{yield}_{avg}$
- $3/7\text{-grass}_{yield} = (0.5 + 2)/3 \times \text{yield}_{avg}$
- $4/8\text{-grass}_{yield} = (0.5 + 3)/4 \times \text{yield}_{avg}$

The energy output was calculated as the product of biomass production and energy content of the harvested biomass, estimated at 18.7 MJ/kg DM (Baxter et al., 2014; Englund et al., 2021).

Crude protein yield was calculated by multiplying DM yield with the average concentration (g/kg DM) of crude protein (i.e., the sum of average fractions A, B_1, B_2, and B_3, see reference) in seven lucerne harvests during field experiments (Solati et al., 2017). True protein was similarly calculated by multiplying DM yield with the average concentration of true protein (i.e., the sum of average fractions B_1, B_2, and B_3) based on the same source.

2.3 | Biogas production and GHG savings from fossil fuel substitution

GHG emissions from biogas production based on biomass from grass production in crop rotations have been estimated at 33 and 30 g CO_2eq MJ^{-1} biogas, with and without upgrading the biogas to natural gas quality, respectively (Börjesson et al., 2015). The estimates were based on the methodology in the EU RED (European Commission, 2009, 2018b) but exclude changes in soil carbon content from grass cultivation and credit for feed output. When upgraded biogas replaces petrol or diesel as transportation fuel in vehicles, the GHG savings are about 61 g CO_2eq MJ^{-1} biogas (a 65% reduction). The reference-fuel lifecycle GHG emissions for petrol and diesel are 94 g CO_2eq MJ^{-1} (European Commission, 2018b). When biogas (not upgraded) replaces natural gas for electricity production, GHG savings are about 38 g CO_2eq MJ^{-1} biogas (a 56% reduction), using reference lifecycle GHG emissions from natural gas of 68 g CO_2eq MJ^{-1} (Prussi et al., 2020). The average methane yield per metric ton DM grass-based feedstock (Börjesson et al., 2015) is 9.2 GJ. Thus, the GHG savings are approximately 560 and 350 kg CO_2eq t^{-1} DM grass when the feedstock is used for biogas production replacing petrol and diesel as vehicle fuel, and natural gas for electricity production, respectively.

2.4 | Effects on soil organic carbon

The effects on SOC from the introduction of the different production systems are based on previous SOC simulations of different agricultural management practices (Lugato et al., 2014). The input data are available for download at the Joint Research Centre European Soil Data Centre (ESDAC; https://esdac.jrc.ec.europa.eu/). The SOC simulation output data are spatially explicit and provide SOC estimates (t C ha^{-1}) for 2010, 2020, 2050, 2080, and 2100, for a business-as-usual scenario (BAU) assuming a continued rotation with the four most dominant crops in each area. They also provide SOC values in relation to BAU for multiple management options, including a grass/annual crop rotation system in which 2 years of lucerne is added to the 4-year BAU rotation. These simulation data were here used as a basis for calculating SOC effects from the different management options and, consequently, the deployment scenarios, as detailed below. Simulation data for a permanent grassland system, in which the BAU rotation is replaced by permanent grassland, are also available. These are here used to assess how SOC increases from the modeled grass/cereal rotations relate to the, assumed, theoretical maximum.

The simulated SOC values were rasterized to match other input data (100 m) and aggregated SOC values were calculated for each landscape by calculating median SOC values. As detailed below, the following information was then calculated for each landscape, management system alternative, and deployment scenario, for 2050, 2080, and 2100, relative to SOC values in 2020 and relative to BAU:

- SOC change per hectare (t C ha^{-1})
- Total SOC (t C)
- Relative SOC change (%)
- SOC change relative to current GHG emissions from agriculture (EEA, 2021).

The simulated SOC values (Lugato et al., 2014) were rasterized to match other input data (100 m) and aggregated SOC values were calculated for each landscape by calculating median SOC values. In the method sequence below, the following codes apply:

- $SOC_{bau\_[year]}$
- SOC BAU values at specific points in time.
- $SOC_{inc\_grass\_[year]}$
- SOC increases relative to BAU from implementation of in-rotation grass systems at specific points in time.
- $SOC_{inc\_permgrass\_[year]}$
- SOC increases relative to BAU from implementation of permanent grassland at specific points in time.
- SOC_{inc} and $SOC_{inc[year]}$
- collectively used below for the above two codes.

SOC_{inc} values are expressed in relation to 2010. They were therefore re-estimated with 2020 as base year, to be able to represent SOC changes from current levels while maintaining 2050, 2080, and 2100 as points in time for assessment. SOC_{bau} did not require re-estimation as it represents a continuation of BAU land use. SOC_{bau_2020} was thus considered representative for current SOC. SOC_{inc} values, however, needed to be re-estimated to represent a 10-year shorter time period than in the original dataset.

To reflect that SOC sequestration tends to be greater in the years following deployment, then declining toward a new equilibrium level as the carbon sink saturates (Smith, 2012). SOC_{inc_2020} was assumed to represent the change in SOC during the first 10 years, that is, between 2020 and 2030:

- $SOC_{inc\_first10} = SOC_{inc\_2020}$

SOC changes during the remaining period (i.e., 20, 50, and 70 years, for 2010, 2080, and 2100, respectively) were calculated by subtracting $SOC_{inc\_first10}$ from $SOC_{inc\_2050|2080|2100}$, thus representing SOC changes in 30|60|80 years following the first 10 years:

- $SOC_{inc\_last30|60|80} = SOC_{inc\_2050|2080|2100} - SOC_{inc\_first10}$

Since SOC changes in 20/50/70 years are required, these values were downscaled by 2/3, 5/6, and 7/8, respectively:

- $SOC_{inc\_last20|50|70} = SOC_{inc\_last30|60|80} \times 2/3|5/6|7/8$

SOC increases by 2050|2080|2100 relative to BAU could then be calculated as:

- $SOC_{inc\_2050|2080|2100} = SOC_{inc\_first10} + SOC_{inc\_last20|50|70}$

These re-estimated SOC values are below referred to as $SOC_{inc\_grass|permgrass\_new\_[year]}$, or collectively as $SOC_{inc\_new\_[year]}$.

At this point, SOC changes by 2050|2080|2100 relative to BAU (t C ha $^{-1}$), with base year 2020, has been identified for the 2/6-grass system. To estimate SOC changes for the other systems, we assumed a linear correlation between SOC changes and the share of total area under annual crops that are used for grass production relative to the 2/6-grass system, on average over time. This approach was selected based on discussions with the developer of the underlying SOC dataset:

- $SOC_{inc\_BAU\_2y\_lim50|lim\_25} = SOC_{inc\_BAU\_2y} \times 0.5|0.25$
- $SOC_{inc\_BAU\_3y|4y} = SOC_{inc\_BAU\_2y} \times 9/7|3/2$

Total SOC changes (t C) were then calculated for each management alternative and in each landscape:

- $SOC_{inc\_total} = SOC_{inc} \times area_{annual\ crops}$

The relative SOC changes (%) could then be calculated for the different assessment years, for example:

- $SOC_{diff\_2y\_2050} = (SOC_{bau\_2050} + SOC_{inc\_2y\_2050}/SOC_{bau\_2050}) - 1$

The same calculations (t C ha $^{-1}$, t C, and %) were also made relative to 2020 instead of BAU. The first was done by adding the difference in BAU SOC between 2020 and the assessment year to the SOC increase relative to BAU for the assessment year, for example:

- $SOCinc_{\_2020\_3y\_2080} = SOC_{inc\_BAU\_3y\_2080} + (SOC_{bau\_2080} - SOC_{bau\_2020})$.

The latter two were calculated as described above. Absolute SOC values for all assessment years and ley systems were then calculated, for example, for the 2/6-grass system by 2050:

- $SOC_{total\_2050} = SOC_{bau\_2020} + SOC_{inc\_2020\_2y\_2050}$

For each production system and assessment year, the share of maximum attainable SOC increase (%) was estimated as the quotient of SOC increase relative to 2020 in the different in-rotation grass systems and in permanent grasslands, respectively, for example:

- $SOC_{inc\_share\_potential\_2y\_2050} = SOC_{inc\_2020\_2y\_2050}/SOC_{inc\_2020\_permgrass\_2050}$

Finally, annual C sequestration relative to total GHG emissions from agriculture (EEA, 2021) (%) was estimated, for example:

- $C\_seq_{total\_2y\_2050\_relBAU} = (SOC_{inc\_total\_2y\_2050}/30)/GHG\_emissions_{current}$
- $C\_seq_{total\_2y\_2050\_rel2020} = (SOC_{inc\_total\_2020\_2y\_2050}/30)/GHG\_emissions_{current}$

2.5 | Environmental co-benefits

Three co-benefits were modeled for each landscape: avoided (i) soil loss by water erosion, (ii) soil loss by wind erosion, and (iii) nitrogen emissions to water. In addition, we indicate potential (iv) mitigated flooding events. Impacts i–iv were quantified for each landscape:

1. Soil loss by water erosion was indicated by "annual average soil loss by water erosion on land used for production of annual crops". Annual soil loss was retrieved from a published dataset for the year 2010 with 100 m resolution (available at ESDAC, see above), based on the application of a modified version of the Revised Universal Soil Loss Equation (RUSLE) model. Average values were then calculated for erosion values on land used for annual crop production, in each landscape.
2. Soil loss by wind erosion, indicated and calculated as for water erosion, based on a 1000-m dataset of soil loss by wind erosion derived using a GIS version (RWEQ-GIS; Borrelli et al., 2017) of the Revised Wind Erosion Equation (RWEQ) model (Fryrear et al., 2000).
3. Nitrogen emissions to water, indicated by "annual average diffuse nitrogen emissions to water," were retrieved by running v2 of the Geospatial Regression Equation for European Nutrient losses (GREEN) model (Grizzetti et al., 2012) for the landscape dataset. Average values were then calculated for erosion values in each landscape.
4. Recurring floods, indicated by "share of landscape area subject to 10-year flooding." Data on 10-year flooding events were retrieved from a published flood hazard dataset with 100 m resolution. The data were derived using a cascading model simulation approach (Alfieri et al., 2014). The share of the total area in each landscape subject to 10-year flooding events was then calculated for each landscape.

The four impacts were classified on a five-step scale from "very low" to "very high." For more details on methods, thresholds, and underlying data, see previous work (Englund, Börjesson, et al., 2020). For impacts 1 and 2, we assumed that the impact is negligible on grassland (Martin et al., 2020). This implies that replacing, for example, 10% of annual crop production with grass would reduce the impact with 10%. The potential impact mitigation in each individual landscape was therefore calculated as the product of the current impact and the share of grassland relative to the current area under annual crops, for the five system designs and the two deployment scenarios. For impact 3, we assumed that nitrogen emissions to water from grass production are 75% lower than current nitrogen emissions to water. This assumption is based on field experiments showing that perennial grasses reduce nitrogen leaching by 70%–80% compared to traditional systems (Manevski et al., 2018). The potential impact mitigation is then calculated in each landscape as the product of current impact, the share of grassland relative to the current annual crop area, and the mitigation factor of 0.75, for the five system designs and the two deployment scenarios. We also estimated to what extent introducing grass production in crop rotations could contribute to reducing impacts 1–3 to a "low" impact level. This was calculated as the quotient of potential impact mitigation by the difference between the upper threshold of the class "low impact" (Englund, Börjesson, et al., 2020) and the current impact.

Flood mitigation could not be estimated using the same approach. There is strong support for claiming that increased grass production in intensively managed agricultural landscapes can mitigate flooding events (Zhao et al., 2020). However, the magnitude of this benefit depends on landscape-specific characteristics and can thus not be generalized in the same way as for the other impacts. We instead attempted to indicate the likelihood of mitigating flooding events as a result of increased grass production in the landscape (Englund et al., 2021). This was done by assuming that the likelihood is directly correlated with the estimated effectiveness of strategic perennialization in mitigating recurring floods (Englund, Börjesson, et al., 2020). A "medium" effectiveness thus corresponds to a "medium" likelihood, etc. The effectiveness of strategic perennialization in mitigating recurring floods was therefore identified for each landscape where the model introduces grass into crop rotations with annual crops.

2.6 | Impacts on crop production

The impact of crop rotation modifications on annual crop production was calculated in three steps:

1. The "direct displacement" effect (%) was calculated as the area under grass cultivation in the two scenarios, respectively, relative to the 91 Mha where grass is introduced in crop rotations (see Section 2.2; Table 1).
2. Land savings due to yield increases (%) were estimated for the 91 Mha above. A yield increase of 20% on all land subject to inclusion of grass in crop rotations was assumed (Marini et al., 2020). Given that some cropland is always under grass cultivation, the total effect on the entire area was calculated as the share of total area under annual crop production in rotation with grass, multiplied by 20. In the high-estimate scenario, 38 Mha (area under grass cultivation on average each year) was subtracted from 91 Mha (total area in rotation) and then divided by 91 Mha (total area), multiplied by 20. In the low-estimate scenario, 15 Mha (area under grass cultivation on average each year) was subtracted from 45 Mha (total area in rotation) and then divided by 91 Mha (total area), multiplied by 20.
3. Potential cropland savings elsewhere were estimated by assuming that the grass biomass is used as feedstock for a protein concentrate that replaces soymeal. Based on FAOSTAT statistics, we assume soybean yields of 2 t ha^{-1} on European cropland and 3.2 t ha^{-1} in Brazil. Furthermore, we assume a moisture content of 14% and that 80% of the soybean is used for soymeal. This results in a soymeal DM yield of 1.34 t ha^{-1} in Europe and 2.20 t ha^{-1} in Brazil. By comparison, grass-based protein concentrate comparable to soymeal can be produced from 14% of the DM grass yield (Corona et al., 2018). In the low estimate, where the average grass yield is 6.8 t DM year^{-1}, this corresponds to 1.16 t protein concentrate ha^{-1}. In the high estimate, where the average grass yield is 7.5 t DM year^{-1}, it corresponds to 1.28 t protein concentrate ha^{-1}. Finally, the potential to replace soybean production (for soymeal) with grass production (for protein concentrate) was calculated as the product of protein concentrate yield (%) and average grass yield (t DM ha^{-1}) in the two scenarios, divided by soymeal yield (t DM ha^{-1}) in Europe and Brazil, respectively.

Steps 1–2 are considered "local effects," and steps 1–3 are considered "net effects."

2.7 | Uncertainties and limitations

Where, and to what extent, implementation takes place, both in the base scenarios and in the high and low estimates, is determined by the thresholds used for classification of impacts and impact mitigation effectiveness (Englund et al., 2021; Englund, Börjesson, et al., 2020). Different thresholds would thus yield different results. General spatial patterns would, however, be similar (Englund,

TABLE 1 Model results for large-scale introduction of grass into crop rotations, aggregated at the European (EU27+UK) scale.

		2/6 years grass	3/7 years grass	4/8 years grass	High-estimate scenario	Low-estimate scenario
Area on which grass is included in annual crop rotations (Mha)		91				45
Average area under grass production (Mha)		30	39	46	38	15
Biomass output (Mt DM year^{-1}|PJ year^{-1})		209|3908	298|5573	365|6826	286|5348	102|1907
Biogas production (PJ year^{-1})		1932	2760	3404	2631	938
Extractable crude protein (Mt)|true protein (Mt)		43|27	62|38	76|47	59|37	21|13
Average SOC increase relative to BAU|relative to 2020 (tC ha^{-1} of total cropland area)	2050	3.2|3.5	4.1|4.4	4.8|5.1	4.1|4.3	1.5|1.9
	2080	4.4|4.9	5.7|6.2	6.6|7.2	5.5|6.0	2.1|2.6
Total SOC increase relative to BAU|relative to 2020 (Mt)	2050	294|335	378|419	442|483	363|404	141|181
	2080	402|476	517|591	603|677	497|570	193|266
Annual GHG emission savings from SOC sequestration until 2050 relative to BAU|relative to 2020 (as % of total current GHG emissions from agriculture)		8.3|9.5	10.7|11.9	12.5|13.6	10.3|11.4	4.0|5.1
Annual GHG savings when biogas substitutes for gasoline and diesel in cars (Mt C year^{-1}|as % of total current GHG emissions from agriculture)		32|27	46|39	56|47	44|37	16|14
Annual GHG savings when biogas substitutes for natural gas for electricity (Mt C year^{-1}|as % of total current GHG emissions from agriculture)		20|17	29|25	35|30	27|23	10|8
Avoided soil loss by water erosion (Mt year^{-1})		76	97	114	95	37
Avoided soil loss by wind erosion (Mt year^{-1})		18	23	27	22	9
Avoided N emissions to water (kt year^{-1})		271	348	406	324	119

Note: BAU = land use continues as per business as usual. Numbers are rounded. See Tables A.1–A.7 for country-level aggregates.

Börjesson, et al., 2020). The use of average simulated yields for miscanthus, switchgrass, and reed canary grass to estimate grass yields is justified by the lack of spatially explicit pan-European yield data for grass/clover species that are traditionally used in rotations with annual crops. Visual assessment of the simulated yields across the study area, based on in-house experience, suggests that reed canary grass yields are the most similar to traditional species, in spatial terms. In absolute numbers, however, miscanthus yields are more similar to what can be expected. Using the average value for these three species provides both reasonable spatial patterns and reasonable yield levels. This approach can be further justified by the fact that selection of grass species is likely to vary across Europe, given different biophysical conditions. It is therefore not reasonable to use simulated yields (if they existed) for one single species, or a specific combination of species, in all landscapes across Europe. See previous studies for general uncertainties related to the underlying models, including co-benefits (Englund et al., 2021; Englund, Börjesson, et al., 2020) and SOC simulations (Lugato et al., 2014). See also Section 4 where model results are evaluated.

3 | RESULTS

The model introduces perennial grass in crop rotations on 91 million hectares (Mha) of arable land, in 24,363 of the ~81,000 assessed landscapes, encompassing about 80% of all land in Europe currently used to cultivate annual crops. Most of these landscapes (76%) are classified as having a "high" level of accumulated SOC losses, 17% are "medium," and 7% are "very high." Adding 2 years of grass cultivation to 4-year crop rotations (2/6-grass system) in these landscapes results in 30 Mha of land being used for cultivation of grass instead of annual crops, on average over time. Adding one additional year of grass in the crop rotation (3/7-grass system) increases the grass area to 39 Mha; adding two additional years (4/8-grass system) results in 46 Mha. The corresponding grass production is about 210, 300, and 370 Mt DM year^{-1}, for the 2/6, 3/7, and 4/8 systems, respectively. The estimated energy content in this biomass is about 4–7 EJ and the corresponding biogas output is about 2–3.4 EJ. Extractable crude protein and true protein amount to about 40–80 Mt and 30–50 Mt, respectively (Table 1).

In our "low-estimate" scenario, the 2/6 system is implemented on all land under annual crop production where SOC loss is classified as "very high", on 50% of the land where it is "high," and on 25% where it is "medium." In this scenario, the total area under grass production amounts to 15 Mha, corresponding to 16% of the area under annual crops in the affected landscapes and 13% of the total area under annual crops in Europe. The corresponding grass biomass production is 100 Mt DM year $^{-1}$, equivalent to an energy content of about 1.9 EJ and a biogas output of 1 EJ. Extractable crude protein and true protein amount to about 20 and 10 Mt, respectively (Table 1).

In our "high-estimate" scenario, the 2/6 system is implemented on all land under annual crop production for which the accumulated SOC loss is classified as "medium," with the 3/7 and 4/8 systems implemented where the loss is classified as "high" and "very high," respectively. Here, the total area under grass production amounts to 38 Mha, corresponding to 41% of the area under annual crops in the affected landscapes and 35% of the total area under annual crops in Europe. The corresponding grass biomass production is 290 Mt DM year^{-1}, equivalent to an energy content of about 5.3 EJ and a biogas output of 2.6 EJ. Extractable crude protein and true protein amount to about 60 Mt and 40 Mt, respectively (Table 1).

In the two deployment scenarios, 70% of the new grass production is established in Poland, Spain, France, Romania, Germany, and Italy. The share of the area under annual crop production devoted to grass is largest in Denmark, Bulgaria, Hungary, Italy, Poland, Greece, Romania, and the Czech Republic (Table A.1; Figure 1).

Effects on SOC are consistently positive (Figure 2; Table 1). If 2, 3, or 4 years of grass are added to a 4-year crop rotation with annual crops, the corresponding SOC increase is about 300, 510, or 600 Mt C by 2080, relative to a business-as-usual scenario in which current land use continues as is (BAU). In the low- and high-estimate scenarios in which implementation depends on the degree of accumulated SOC losses, the total SOC increase by 2080 is 190 and 500 Mt C, respectively. There are, however, substantial variations between different regions and individual landscapes. For example, while the average landscape in the high-estimate scenario achieves an increase in SOC (by 2080) of 5.1 t ha $^{-1}$, the 20th percentile is 2.2 and the 80th percentile is 6.3 t ha $^{-1}$.

In the high-estimate scenario, the total average annual SOC sequestration by 2050 amounts to 12.1 Mt C year^{-1} relative to BAU; in the low estimate, it amounts to 4.7 Mt C year^{-1}. This is equivalent to about 4%–10% of the total current GHG emissions from agriculture in EU27 + UK (EEA, 2021). Comparing with 2020 levels instead of BAU results in slightly higher values. The combined GHG savings from increases in SOC and from decreases in fossil fuels due to an increased use of biogas amount to 13%–48% of current GHG emissions from agriculture. The range depends on the deployment scenario, whether biogas displaces natural gas in power plants or is upgraded to vehicle fuel displacing petrol and diesel in cars, and whether SOC increases are estimated relative to BAU or 2020 levels, see Table 1.

Bulgaria, Romania, Belgium, Slovakia, and Hungary have the greatest average SOC increase in the two deployment scenarios. Finland, Estonia, Slovenia, and Sweden have the lowest. In total, 80% of the modeled SOC increase takes place in France, Romania, Poland, Denmark, Italy, Spain, Hungary, and Bulgaria (Figure 2; Tables A.2–A.4).

3.1 | Co-benefits

The other relevant environmental problems differ in magnitude across Europe (Figure 3; see also previous work; Englund et al., 2021; Englund, Börjesson, et al., 2020). For example, nitrogen emissions to water are high in the northwest and central parts of Europe. Water erosion is primarily a problem in the southern and central parts. Wind erosion is primarily a problem in coastal areas in northern and eastern Europe, and recurring floods are problematic all over Europe, mainly around major rivers. While all these problems could theoretically be mitigated by growing more grass, the mitigation potential is, naturally, determined by the location and magnitude of the problem (Figures 3 and 4).

In the low-estimate scenario, nitrogen emissions to water decrease by a total of 119 kt N year^{-1}; in the high-estimate scenario, the figure is 324 kt N year^{-1} (Table 1). In the low-estimate scenario, grass rotations contribute 34% of the reduction necessary to reduce the impact down to a "low" level, in the median landscape. In the high-estimate scenario, the same contribution surpasses 100%.

A substantial mitigation potential is also seen for soil loss by water erosion, which is reduced by 37 and 95 Mt annually in the low- and high-estimate scenarios, respectively (Table 1). For the median landscape, this translates into 33% of the reduction necessary to reach the "low" impact level in the low-estimate scenario and 85% in the high-estimate scenario.

Soil loss by wind erosion is generally a smaller problem, but the mitigation potential is nevertheless substantial in areas where it is severe. The total reduction potential is 9 Mt and 22 Mt year^{-1} in the low- and high-impact scenarios, respectively (Table 1). For the median landscape, this corresponds to 48% of the reduction necessary to reach the "low" impact level in the low-estimate scenario. In the high-estimate scenario, the reduction surpasses 100%.

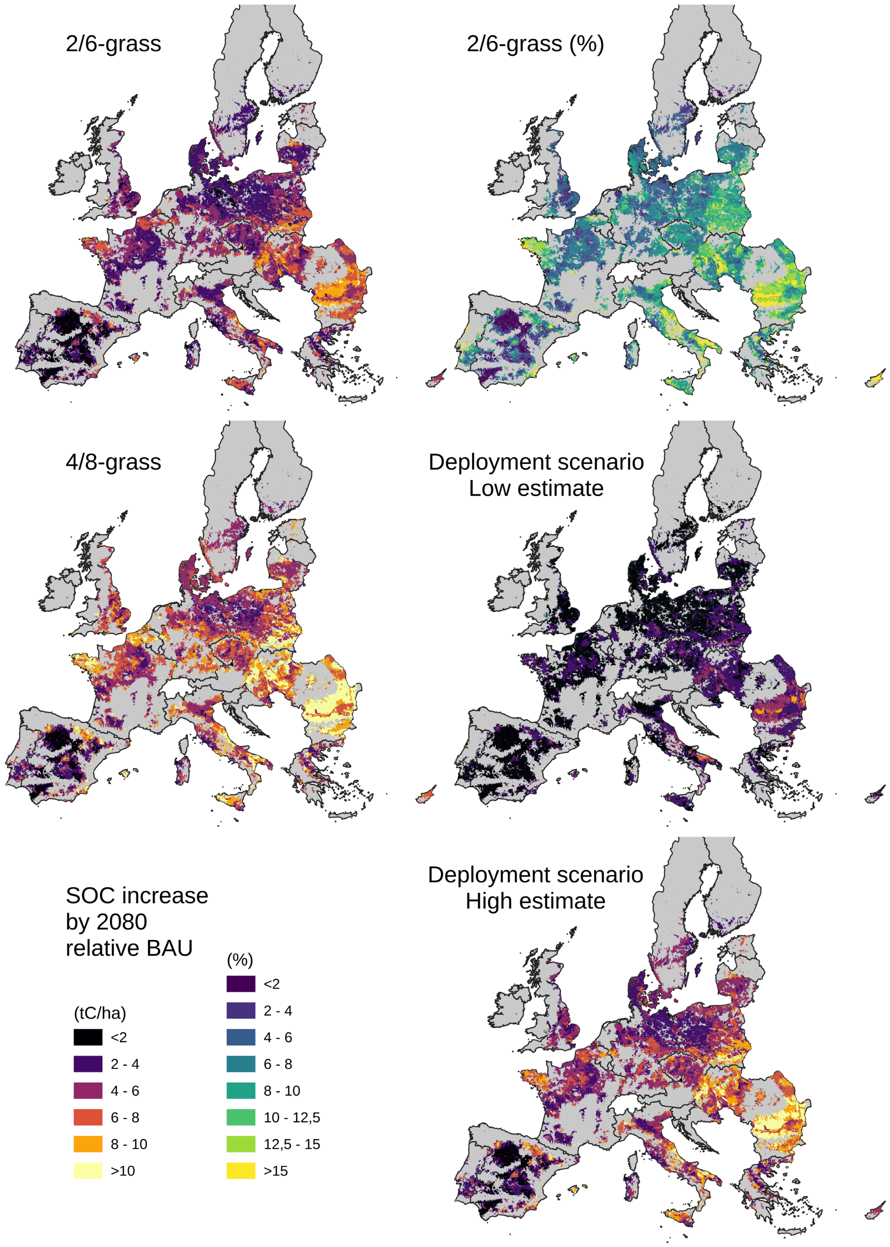

FIGURE 2 Increase in soil organic carbon (SOC) from the introduction of grass in crop rotations, relative to a business-as-usual (BAU) scenario.

The co-benefits are thus considerable. In the high-estimate scenario, no further measures are needed to reduce nitrogen emissions to water nor soil loss by wind erosion in most landscapes where grass production is included in annual crop rotations. In addition to the co-benefits described above, there are multiple other co-benefits that are possible, and even likely, that have not been quantified, such as a reduced need for pesticides. Furthermore, mitigated flooding events have not been modeled explicitly, but an indicative assessment shows that the likelihood of mitigated flooding events is classified as "medium" in 12% of the landscapes where grass is included in the rotation, "high" in 13%, and "very high" in 3% (Figure 5). Potential additional co-benefits thus need to be better understood and quantified to get a more complete picture of the positive effects of large-scale deployment of grass production in crop rotations.

3.2 | Impacts on crop production

When grass is introduced via adjustments to crop rotations on 91 Mha of cropland used for annual crop cultivation, the area cultivated with annual crops every year is reduced to 53 and 76 Mha in the high- and low-estimate scenarios, respectively (Table 1 and Figure 6). When factoring in yield increases obtained when the annual crops are cultivated in rotation with grass (see Section 4), the cropland displacement effect is reduced. The need for additional annual crop cultivation area outside the 91 Mha, to maintain the total annual crop production at the level prior to crop

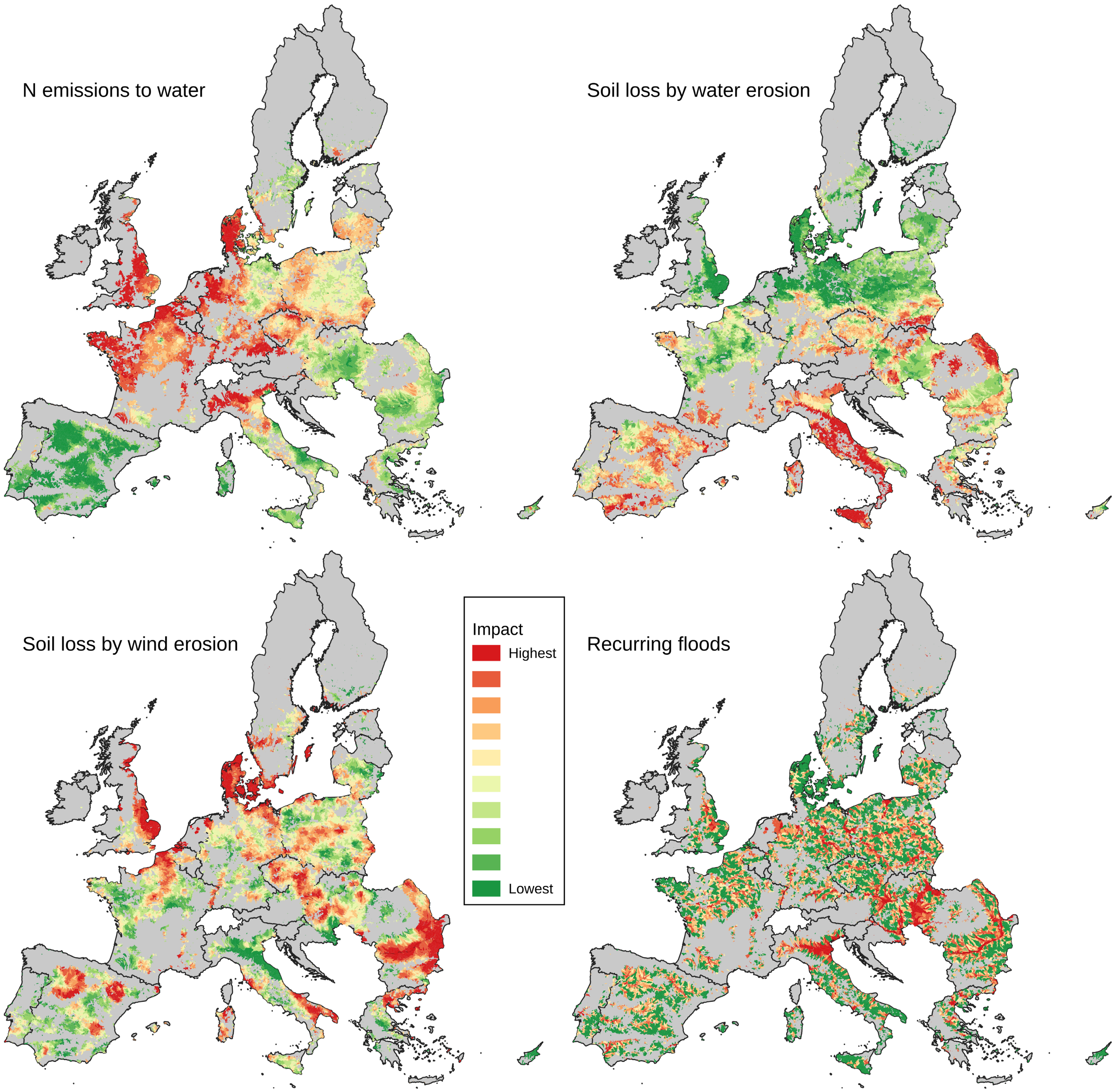

FIGURE 3 Current degree of N emissions to water, soil loss by wind erosion, soil loss by water erosion, and recurring floods, in the landscapes where grass is introduced in crop rotations.

rotation adjustments, is then 9 and 27 Mha in the high- and low-estimate scenarios, respectively (Figure 6).

If the grass biomass is processed into products previously produced from annual crops, the cropland displacement effect is further reduced. Considering only the soymeal substitution effect, 1 ha of grass cultivation can substitute 0.84–0.93 ha of European soybean cultivation, or 0.53–0.58 ha of Brazilian soybean cultivation (low–high estimate).

Thus, the net cropland displacement effect of crop rotation adjustments can be relatively small, even negative (i.e., net cropland savings); it ranges from 2% to −3% in the low-estimate scenario and from 8% to −5% in the high-estimate scenario.

4 | DISCUSSION

The results show that there is a substantial SOC sequestration potential on European cropland when adding 2–4 years of grass to a 4-year rotation with annual crops (for a total rotation of 6–8 years) at a large scale. The

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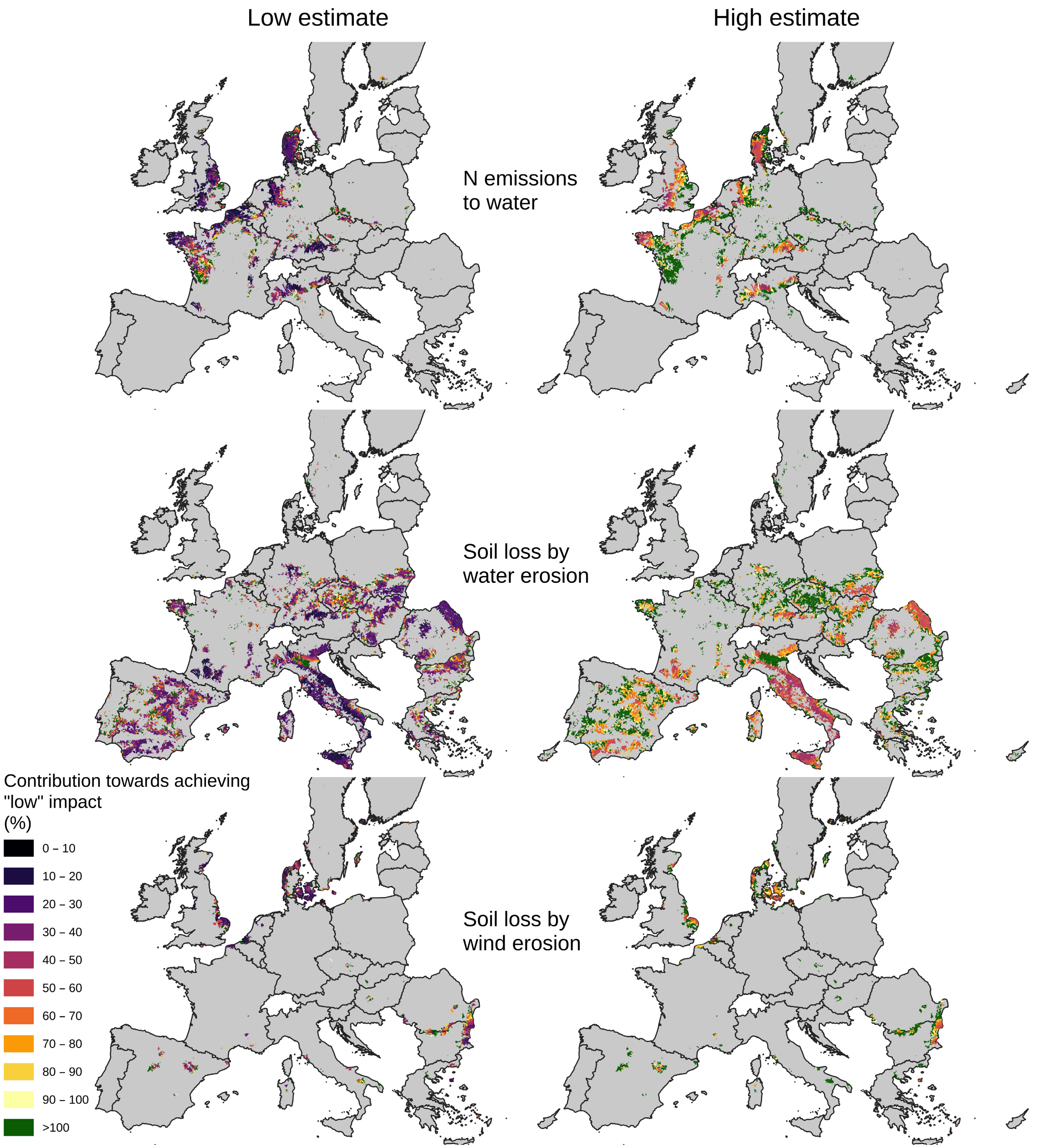

FIGURE 4 Co-benefits of introducing grass production in crop rotations with the primary objective of enhancing soil organic carbon. The figure shows the relative to contribution toward reaching the classification "low impact" at the landscape scale for nitrogen emissions to water, soil loss by water erosion, and soil loss by wind erosion, respectively, in the low-estimate (left) and high-estimate (right) scenarios. Landscapes that already have a "low" or lower impact are excluded.

environmental co-benefits (reduced wind and water erosion and reduced nitrogen emissions to water), are also substantial—in some cases exceeding the estimated mitigation needs. Our results also indicate possible mitigation of flooding events all across Europe. The results are in line with (and based on) previous research on SOC effects from grass production in rotation with annual crops (Lugato et al., 2014), but since there are, to our knowledge, no previous studies that spatially link grass production to multiple ecosystem services at European scale, the results

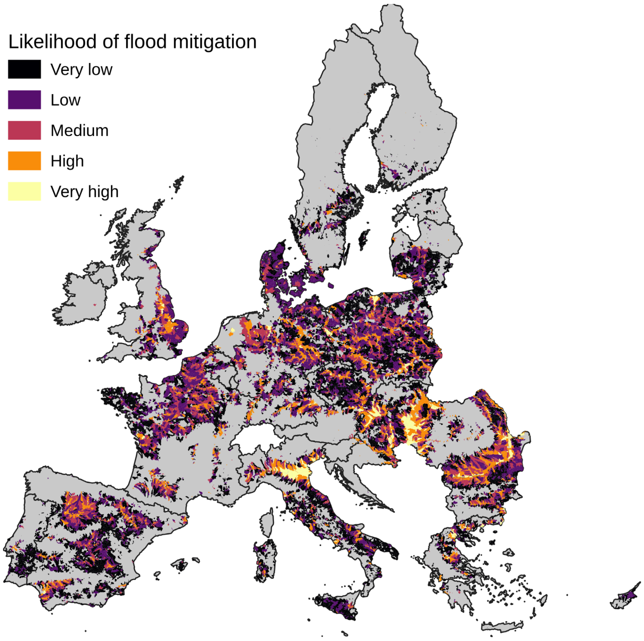

FIGURE 5 Likelihood of mitigated flooding events as a result of widespread deployment of grass in crop rotations. Note that this is a general indication of how problems with flooding in a landscape can be mitigated by increased cultivation of perennials, and that there is no distinction made between the different deployment scenarios.

are otherwise difficult to evaluate. The model is based on a combination of several large-scale models that all have their limitations and uncertainties (see Section 2). Also, farm management and local conditions will affect the outcome of implementing the assessed production systems, in ways that cannot be captured by pan-European models.

While the input SOC data are based on a validated model (Lugato et al., 2014), it is important to also evaluate the output. For this purpose, measurements from long-term agricultural field trials are valuable, albeit scarce. In England, SOC changes have been measured since 1938, when an arable 5-year rotation with cereals and root crops was changed into a rotation with 3 years of grass and 4 years of cereal crops (i.e., a 60% share of grass in the overall rotation, cf. Table 2). The measurements over 70 years reveal an average annual SOC sequestration in the topsoil (0–25 cm) of 0.34 tC ha^{-1} year^{-1} during the first 30 years and 0.15 tC ha^{-1} year^{-1}, thereafter (Johnston et al., 2017). Börjesson et al. (2018) report long-term field measurements for two sites in southern Sweden with different climate and soil characteristics. Here, a 4-year rotation with cereals was changed into a mixed rotation with 3 years of grass and 1 year of cereals around 1980. After 35 years, significant increases in SOC concentrations and stocks were found in the grass-dominated rotations compared with cereal monoculture, 0.36–0.59 tC ha^{-1} year^{-1} (topsoil, 0–20 cm). The results reported in this study (Table 2) appear to be conservative compared with these field trials. The results also illustrate that SOC sequestration increases with the share of grass in the total crop rotation (Christensen et al., 2009; Jarvis et al., 2017) and confirm that SOC sequestration tends to be greater in the years following deployment, then declining toward a new equilibrium level as the carbon sink saturates (Smith, 2012).

The combined GHG savings from increases in SOC and fossil fuel substitution by biogas amount to 13%–48% of current GHG emissions from agriculture in Europe. This estimate does not consider potential increases in N_2O emissions due to incorporation of residues in soil, which depend on how the biomass is treated. If harvested and removed (e.g., as feedstock to biorefineries and/or anaerobic digesters), a small amount of above-ground residues are left in the field, and only below-ground residues, that is, nitrogen in root systems, will contribute to N_2O emissions (IPCC, 2019b). Recent research indicates that these effects are negligible, primarily due to reduced fertilizer needs (Lugato et al., 2020). Furthermore, grass cultivation for more than one single year reduces the time with bare soil and thus increases surface albedo compared with conventional annual cropping systems. This reduction in albedo-driven radiative forcing provides additional and more immediate climate benefits (Lugato et al., 2020).

Biodiversity is rapidly declining (IPBES, 2019). One important cause is the extensive use of insecticides and fungicides, which consistently have negative effects on biodiversity and on the potential for biological pest control (Geiger et al., 2010). Grass production in crop rotations has a very low (or zero) need for pesticides, especially fungicides and insecticides (Nordborg et al., 2017). Including grass in crop rotations with annual crops would thus reduce the overall need for pesticides and consequently reduce impacts on biodiversity from agriculture (Urruty et al., 2016). Increased crop diversity is also an important measure to increase biodiversity at the landscape level (Sirami et al., 2019).

The introduction of grass/legume species into annual crop rotations reduces the harvested area of cereal crops. This cropland displacement impacts food/feed production and may counteract environmental benefits, including reduced pesticide use, by causing cropland intensification or expansion elsewhere. However, this effect can to some extent be counterbalanced (Figure 6). Changes to more diversified crop rotations are well known to enhance the yield of grain crops, such as wheat. The principal mechanisms behind these yield gains include enhanced disease control and improved supply of nitrogen and water. There are, however, other "rotation effects" that are not yet fully understood (Kirkegaard et al., 2008). Wheat yields

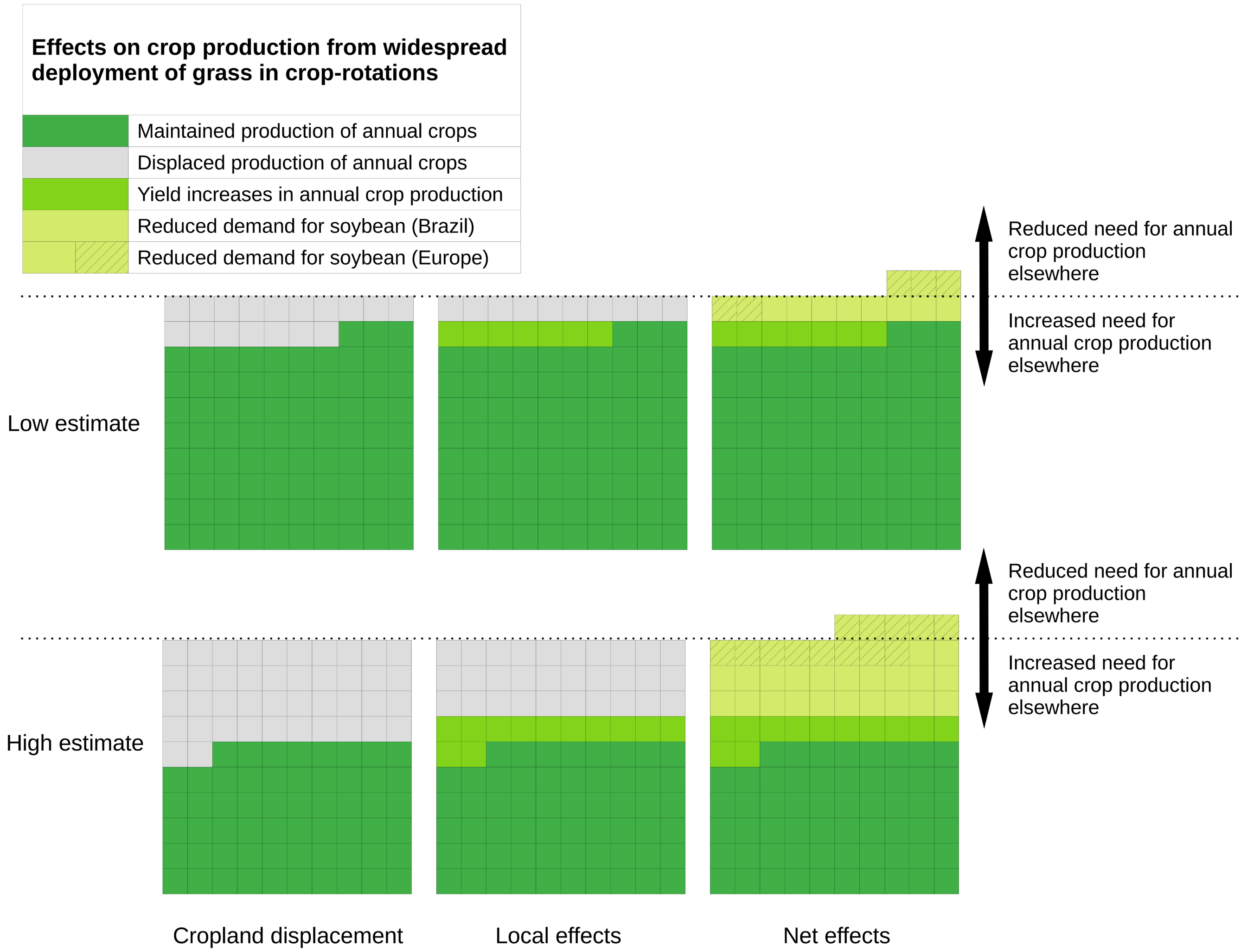

FIGURE 6 Effects on the production of annual crops due to widespread deployment of grass in crop rotations, compared to a reference case with annual crops in monoculture. Each cell corresponds to current production of annual crops on 91/100 Mha.

TABLE 2 Average annual soil organic carbon sequestration rate (tC ha^{-1} year^{-1}) in the three systems where grass is included in rotation with annual crops.

System (year grass/year total rotation|% grass)	Average annual SOC sequestration rate (tC ha^{-1} year^{-1}) 30 years	60 years
2/6|33%	0.11	0.07
3/7|44%	0.14	0.1
4/8|50%	0.16	0.11

preceded by a break crop have been shown to increase from 0.5 t ha^{-1} (pre-crop: oats) to 1.2 t ha^{-1} (pre-crop: grain legumes) compared to when preceded by wheat (Angus et al., 2015), corresponding to 12%–29% of the average wheat yields in 2020. The effect in the second wheat harvest after a break crop corresponds to 20%–60% of that in the first year (Angus et al., 2015). As with SOC sequestration rates, confirming overall rotation effects on yields requires data from long-term agricultural field trials. In an analysis from seven such trials across Europe with consecutive yield data for time periods ranging 20–55 years, Marini et al. (2020) show that diversified crop rotations including 2–3 years of grass/legumes in overall 6- to 7-year rotations provided higher yields for both winter and spring cereals (on average +0.86 and +0.39 t ha^{-1} year^{-1}, respectively), compared with a continuous monoculture of cereals. The yield gains were higher, up to around 1 t ha^{-1} year^{-1}, in years with high temperatures and limited precipitation (Marini et al., 2020). Diversifying crop rotations thus appears to be an interesting adaptation measure under a changing climate. Angus et al. (2015) estimate that at the global level, 40% of the wheat area is not preceded by an effective break crop, forage, or fallow, indicating a substantial potential for yield increases. In the EU, cereals (primarily wheat) dominate among crops on arable land, but estimates of the potential yield increases from diversified crop rotations are lacking.

The food/feed crop displacement effect is further reduced when grass biomass is used in biorefineries that can produce food or feed along with bioenergy and other bio-based products (Aristizábal-Marulanda & Alzate, 2019; Njakou Djomo et al., 2020; Schmidt et al., 2019). For example, lactic acid bacteria can facilitate the use of grass biomass for the production of a protein concentrate, suitable for feeding monogastric animals as well as ruminants, with multiple co-products (Lübeck & Lübeck, 2019). Trials in Denmark show that grass protein with a high protein content (47% DM) can substitute for soymeal in pig feed without any adverse effects on animal performance or meat quality (Santamaría-Fernández & Lübeck, 2020). Such solutions using alternatives to high-input and high-emission annual grain and seed crops as feedstock can enable sustainable intensification of agricultural systems with reduced environmental impacts (Larsen et al., 2017).

One option for mitigating crop displacement effects is to prioritize grass rotations on degraded or low-productivity cropland, while prioritizing annual crop production on high-productivity lands. While this has not been explicitly explored in this study, it relates to what the two scenarios entail—that grass production is targeted to cropland with the greatest accumulated SOC losses and, consequently, to land where the yields are most negatively affected by historic land use and cultivation of annual crops can be(come) economically challenging. On such lands, grass cultivation may be particularly valuable in helping to improve soil quality and enhancing long-term economic viability of annual crop cultivation.

Beyond mitigation of cropland displacement, protein feed production in Europe can substitute for imported plant protein, mostly soymeal, which is a major import commodity to the EU food sector, both in terms of volume and use of agricultural land abroad (Poux & Aubert, 2018). Since this import is associated with substantial environmental concerns (deforestation, biodiversity loss, extensive pesticide use, etc.), the motives for developing a substitute source of feed protein are strong (Santamaría-Fernández & Lübeck, 2020). This is highlighted by recent efforts by the European Commission to support EU-grown plant-based protein use, via support schemes in the new CAP and by boosting innovation and technology development (European Commission, 2018a). Furthermore, the increased target goal in the recent proposal for a revision of the EU Renewable Energy Directive (European Commission, 2021), where the share of renewable energy should amount to 40% in 2030, is likely to be a strong driver for increased production of biogas for heat, power, and transportation fuel. Here, the outcome of the current process following the European Commission's proposal (European Commission, 2021) to revise the Renewable Energy Directive (RED) will likely influence how investors consider biorefineries. For example, treatment of biogas from biorefineries in the revised RED will depend on whether biogas is considered a main product or co-product of the biorefinery process.

A prerequisite for widespread deployment of grasses in crop rotations is a demand for products that can be produced from the grass biomass (Englund, Dimitriou, et al., 2020), although some farmers may consider soil quality improvements sufficient motivation. Grass cultivation may also be an attractive option where intensive annual crop cultivation becomes restricted to protect the environment. In other places, incentives such as payments for soil carbon sequestration and other environmental benefits may be needed (Englund, Dimitriou, et al., 2020). Such payment schemes require reliable methods for quantifying environmental effects with high detail, within individual landscapes.

Biomass cultivation systems are connected to, and interact with, surrounding and supporting systems, for example, the soil system and adjacent landscapes. Such interactions are not well captured in environmental assessments conducted based on life cycle assessment (LCA). This is partly because the product-based approach followed by this method focuses on the output of specific provisioning services, and partly because key aspects of sustainable agriculture, for example, better soil health, lower biodiversity impacts, and lower pesticide-use impacts, are generally ignored (van der Werf et al., 2020). Spatial modeling, such as in this study, can provide complementary information about biomass cultivation systems, including their output in terms of provisioning, maintaining, and cultural ecosystem services. Spatial modeling can support assessment of multiple environmental effects from different land-use scenarios over a large geographic area while quantifying effects at different aggregation levels and providing spatially explicit details at multiple scales. However, a large geographic area typically comes with a loss of precision, as local conditions cannot be fully considered (Englund, Börjesson, et al., 2020). Attention to more detailed landscape-level analyses (Busch, 2017; Ssegane et al., 2015) is thus needed to understand how to optimize conditions for biodiversity and multiple ecosystem services, while upholding biomass production.

Finally, achieving substantial environmental benefits and net cropland savings, as illustrated in this article, requires that grass biomass replaces annual crops as raw material for bio-based products. If the grass biomass is instead used to produce new products, for example, to displace fossil fuels, the need to produce additional annual crops elsewhere will increase. Given the uncertainty regarding future policy objectives and their effects on markets for grass biomass and environmental benefits, the actual effects on the global, or even European, agricultural system depends on many factors and transcend regions as well as continents. Complementary studies, such as integrated assessment modeling, can provide important additional insights about the consequences of widespread deployment of grass cultivation, on food and feed production and markets, energy systems and security, GHG emissions, and environmental benefits.

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CONFLICT OF INTEREST

The authors declare no conflicts of interest.

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