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Beneficial land use change: Strategic expansion of new biomass plantations can reduce environmental impacts from EU agriculture

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ABSTRACT

Society faces the double challenge of increasing biomass production to meet the future demands for food, materials and bioenergy, while addressing negative impacts of current (and future) land use. In the discourse, land use change (LUC) has often been considered as negative, referring to impacts of deforestation and expansion of biomass plantations. However, strategic establishment of suitable perennial production systems in agricultural landscapes can mitigate environmental impacts of current crop production, while providing biomass for the bioeconomy. Here, we explore the potential for such "beneficial LUC" in EU28. First, we map and quantify the degree of accumulated soil organic carbon losses, soil loss by wind and water erosion, nitrogen emissions to water, and recurring floods, in ~81.000 individual landscapes in EU28. We then estimate the effectiveness in mitigating these impacts through establishment of perennial plants, in each landscape. The results indicate that there is a substantial potential for effective impact mitigation. Depending on criteria selection, 10–46% of the land used for annual crop production in EU28 is located in landscapes that could be considered priority areas for beneficial LUC. These areas are scattered all over Europe, but there are notable "hot-spots" where priority areas are concentrated, e.g., large parts of Denmark, western UK, The Po valley in Italy, and the Danube basin. While some policy developments support beneficial LUC, implementation could benefit from attempts to realize synergies between different Sustainable Development Goals, e.g., "Zero hunger", "Clean water and sanitation", "Affordable and Clean Energy", "Climate Action", and "Life on Land".

1. Introduction

The exploitation of fossil fuels has been a powerful driver of global societal development in the twentieth century, resulting in a reduced relative dependency on biomass. One notable example is the complete transformation of the energy systems — from biomass based to fossil based. The food sector has also undergone large changes; while most of our food still comes from agriculture, it is often produced in an intensive manner, relying on fossil fuels and petroleum-based chemicals. This development, especially the invention of synthetic fertilizers, has limited the need for expanding agricultural land, while the global population, and its affluence, has steadily increased. Nevertheless, biomass resources are of major significance for the economy in many countries (FAO, 2014; Alston and Pardey, 2014). As a growing and wealthier global population requires more food, paper, construction wood, and other biomaterials, the demand for land and biomass is expected to increase (Scarlat et al., 2015). This is further accelerated by societal concerns about resource scarcity and impacts associated with the use of non-renewable resources — not the least climate change (Scarlat et al., 2015). Visions of a biobased circular economy have caused countries, organizations, and companies to adopt policies, regulations, and strategies aimed at substituting fossil materials with biomass (D'Amato et al., 2017). Most notably, bioenergy is expected to play a major role in the substitution of fossil energy necessary to meet

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global climate targets (Clarke et al. (2014); IPCC, 2018).

At the same time, human societies have already put almost half of the world's land surface to their service and have caused extensive land degradation and loss of biodiversity worldwide (Rockström et al., 2009). As we manage landscapes and associated ecosystems for the production of biomass, we often alter their capacity to support other ecosystem services (ES) that are essential for human well-being (Smith et al., 2013). Many ecosystems are currently being degraded or used unsustainably, jeopardizing their capacity to support multiple ES over time (Costanza et al., 2014). The cultivation of annual crops is an important example, as, e.g., nutrient and agrochemical runoff to water bodies, soil carbon losses, and erosion can cause impacts such as eutrophication, climate change, and soil degradation, in the absence of a parallel supply of ES (i.e., nutrient retention, soil carbon sequestration and regulation of mass flows) that can regulate these stressors (Power 2010). Such impacts can be observed in all parts of the world where there is intensive production of annual crops, including Europe (Grizzetti et al., 2012; Panagos et al., 2015; Borrelli et al., 2017; Alfieri et al., 2014; Lugato et al., 2014a).

Implications of an increased biomass supply have therefore been debated for many decades, primarily focusing on bioenergy, with key issues being land use impacts and uncertain climate benefits (Abad et al., 2017; Berndes et al., 2003; Creutzig et al., 2015; Leemans et al., 1996; Slade et al., 2014; Smith et al., 2013). One example is the debate and research activity following the biomass intensive scenario (LESS) in the Second Assessment Report of IPCC. More recently, a similar debate has arisen following IPCC AR5 (Clarke et al., 2014) and IPCC SR1.5 (IPCC, 2018), in which bioenergy with carbon capture and storage is relied upon in most of the considered scenarios where the mean temperature increase is limited to 1.5 °C or 2 °C above the pre-industrial level. In the discourse, land use change (LUC) has often been considered as negative, referring to environmental and socio-economic impacts of deforestation and expansion of biomass plantations on previously uncultivated land, e.g., habitat loss, greenhouse gas emissions, soil degradation, and water pollution (Searchinger et al., 2008; Kline and Dale 2008; Berndes et al. 2012). In relation to the IPCC AR6 cycle, Smith and Porter (2018) identify key emerging issues to be (i) trade-offs between the use of land for bioenergy production, food and fibre production, and conservation of ecosystem integrity and (ii) the codelivery of bioenergy based climate change mitigation (with or without carbon capture and storage) and the UN Sustainable Development Goals (SDGs). These issues were also prominent in the recently approved Summary for Policymakers of the IPCC report on Climate Change and Land (IPCC 2019).

Society thus faces the double challenge of increasing biomass production to meet the future demands for food, materials and bioenergy, while addressing negative impacts of current (and future) land use. In relation to this, there is a growing body of literature that investigates opportunities for achieving ``beneficial LUC", where a strategic integration of perennial plants (``perennials") into agricultural landscapes enhances, e.g., landscape diversity, habitat quality, retention of nutrients and sediment, erosion control, climate regulation, pollination, pest and disease control, and flood regulation (Asbjornsen et al., 2014; Berndes et al., 2008; Christen and Dalgaard, 2013; Dauber and Miyake, 2016; Holland et al., 2015; Milner et al., 2016; Styles et al., 2016; Ssegane et al., 2015; Ssegane and Negri, 2016; Zumpf et al., 2017; Cacho et al., 2017). Such LUC can thereby mitigate environmental impacts from intensive agriculture, while maintaining or increasing total productivity. Perennial grasses (e.g. Miscanthus, reed canary grass, switchgrass) as well as woody plants (e.g., short-rotation coppice willow or poplar) can be used for such purposes. There is significant experience of this type of biomass supply systems from both practical field trials and commercial applications (Berndes et al., 2008; 2004; Börjesson, 1999a; Börjesson and Berndes, 2006; Christian et al., 1994; Göransson, 1994; Grigal and Berguson, 1998; Gustafsson, 1987; Kort et al., 1998; Perttu and Kowalik, 1997; Rijtema and DeVries, 1994). Implementation of beneficial LUC through such strategic perennialization can support a growing use of bioenergy and other bio-based products while advancing several SDGs, e.g., ``Zero hunger", ``Clean water and sanitation", ``Affordable and Clean Energy", ``Climate Action", and ``Life on Land".

Most earlier studies of beneficial LUC, as referred to above, are conceptual or adopt a limited geographical scope. Few have investigated the possible extent and spatial distribution at larger scales. This article presents the first attempt to explore the potential for beneficial LUC across EU28, based on high-resolution land use modeling. We identify and quantify:

(1) The degree of selected environmental impacts associated with agriculture (soil loss by wind and water erosion, nitrogen emissions to water, accumulated loss of soil organic carbon (SOC), and recurring floods) in ~81 000 individual landscapes in EU28.
(2) The extent to which strategic introduction of perennials in individual landscapes (from here on referred to as ``strategic perennialization") could mitigate these impacts.
(3) Agricultural areas where strategic perennialization may be particularly beneficial from an environmental point of view.

Finally, we discuss policy implications for realizing beneficial LUC on a larger scale in EU28.

2. Material and methods

2.1. Spatial analysis unit

The spatial analysis unit for the assessment is equivalent to functional elementary catchments (FECs) from the ECRINS database (European Environment Agency, 2012), modified as specified below. FEC is equivalent to sub-watershed. This unit was selected based on the importance of hydrological processes, constrained by a watershed, in determining how nutrient and sediment retention and the control of water and mass flows can be affected by a change in land use. It was also considered an appropriate size for assessing implementation options.

Throughout this article, the analysis units are also referred to as *landscapes*. While there are varying meanings of the term landscape, it is here defined as an intermediate integration level between the field and the physiographic region (Burel and Baudry, 2003; Turner, 1989), with an extent depending on the spatial range of the biophysical and anthropogenic processes driving the processes under study (Lacoste et al., 2014). The term “landscape-scale” is also commonly used in both scientific studies and policies concerning implementation of measures for mitigating environmental impacts (Englund et al., 2017). A thorough discussion on the use of the terms *landscape* and *landscape scale* is provided by Englund et al. (2017).

2.1.1. FECs to landscapes

The following modifications were made to the original FEC dataset.[1] All GIS operations were made using the coordinate reference system ETRS89-LAEA Europe (EPSG:3035).

1 The original dataset included a total of 81,301 FECs in EU28, Norway, and Switzerland. In the construction of the original dataset (European Environment Agency, 2012), a number of FECs were represented by more than one polygon. This had to be resolved since one landscape cannot consist of several polygons. These multi-polygon FECs could not be “dissolved” since they in many cases were not located next to each other. Instead, they were split into

[1] In the below description, “FEC” refers to features in the original dataset while “landscape” refers to features in the resulting dataset.

multiple individual polygons. This increased the number of features to 95,086.

2 Original FECs enveloped waterbodies. It was considered more appropriate to consider landscapes as land units. Also, it was observed that large lakes were split between several different surrounding FECs, which is unrealistic. To resolve this, a lake dataset from the ECRINS project (European Environment Agency, 2012) was used to exclude all lakes from the landscape dataset. This increased the number of features to 115,804.

3 In the construction of the original FEC dataset, many very small polygons were created, e.g., in FEC intersections. At the time, the complexity of correcting this was considered to outweigh the benefit (European Environment Agency, 2012). An effort was therefore made here to delete polygons that could be considered noise, to avoid unrealistic quantifications of the results. This was done by deleting all 26,560 features smaller than 5 ha (of which 12,366 features < 1 ha, 10,988 = 1 ha, 1729 = 2 ha, 922 = 3 ha, and 575 = 4 ha), constituting less than 0.01% of the total study area. The threshold of 5 ha was determined based on visual inspection of randomly selected features of different sizes. Although this operation may have resulted in the removal of a few “actual” landscapes, e.g., very small islands, from the dataset, the benefit of reliable quantifications were considered to outweigh this. This decreased the number of features to 89,244.

4 Finally, all landscapes in Norway (6645) and Switzerland (1127) were deleted from the dataset, in order to only consider landscapes subject to the Common Agricultural Policy (CAP) regulations of the EU. The remaining and final number of landscape units was then 81,472.[2]

2.2. *Degree of negative environmental impacts*

Five environmental impacts that could be mitigated by the introduction of perennials into intensive arable landscapes were included in this assessment (Table 1). Each impact can be attributed to insufficient supply of, or degraded, ES under current agricultural practices. The relationship between ES, environmental impacts, and the spatial indicator used for impact classification is available in Table 1.

Each landscape was classified as having *very low, low, medium, high*, or *very high* (i) nutrient emissions to water, (ii) soil loss by water erosion, (iii) soil loss by wind erosion, (iv) recurring floods, and (v) accumulated loss of soil organic carbon (SOC). This classification was made using spatial indicators, as summarized in Table 1 and described below.

2.2.1. *Nitrogen emissions to water*

Indicated by ``annual average diffuse nitrogen emissions to water'', retrieved by running v2 of the Geospatial Regression Equation for European Nutrient losses (GREEN) model (Grizzetti et al., 2012) for the landscape dataset. Diffuse sources include mineral fertilizers, manure applications, atmospheric deposition, crop fixation, and scattered dwellings. For each sub-basin (i.e., landscape), the model considers the total input of diffuse sources and estimates the nutrient fraction retained during the transport from land to surface water.

Thresholds for classification were based on expert (i.e., model developer) recommendations (Table 1).

2.2.2. *Soil loss by erosion*

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 the Joint Research Centre European Soil Data Centre (ESDAC; https://

[2] This was done as step 4 instead of step 1 due to an initial aim of including these countries in the assessment.

Table 1
Spatial indicators, units and thresholds for classifying landscapes as having varying degrees of negative environmental impacts, and the corresponding ecosystem service that is required for impact mitigation.

Environmental impact	Ecosystem service required for mitigation	Spatial indicator	Unit	Degree of environmental impact				
				Very low	Low	Medium	High	Very high
Nitrogen emissions to water	Nutrient retention	Annual average diffuse nitrogen emissions to water	kg N / ha / y (Average value for entire landscape)	≤5	(5,10]	(10,15]	(15,20]	> 20
Soil loss by water erosion	Mass flow regulation	Annual average soil loss by water erosion on land used for production of annual crops	t soil loss / ha / y (Average value on land used for production of annual crops)	0	(0,2]	(2,5]	(5,10]	> 10
Soil loss by wind erosion	Mass flow regulation	As above, but for wind erosion	as above.					
Recurring floods	Water flow regulation	Share of landscape area subject to 10-year flooding	% of landscape area	0	(0,5]	(5,10]	(10,25]	> 25
Accumulated loss of soil organic carbon (SOC)	Soil organic matter formation and composition	Average SOC saturation capacity on land used for production of annual crops	Ratio of current SOC divided by theoretical max SOC (Average value on land used for production of annual crops)	> 0.844 + null	(0.688, 0.844]	(0.532, 0.688]	(0.376, 0.532]	≤0.376
Accumulated loss of SOC – low estimate (−10%)				> 0.7596 + null	(0.6192, 0.7596]	(0.4788, 0.6192]	(0.3384, 0.4788]	≤0.3384
Accumulated loss of SOC – high estimate (+10%)				> 0.9284 + null	(0.7568, 0.9284]	(0.5852, 0.7568]	(0.4136, 0.5852]	≤0.4136

esdac.jrc.ec.europa.eu/) based on the application of a modified version of the Revised Universal Soil Loss Equation (RUSLE) model (RUSLE2015), within which rainfall erosivity, soil erodibility, cover-management, topography, and support practices were modelled with the most recently available pan-European datasets (Panagos et al., 2015).

The degree of soil loss from wind erosion was estimated and classified as described above for water erosion, but using published data with 1000 m resolution (available at ESDAC). The data were derived using a GIS version (RWEQ-GIS) (Borrelli et al., 2017) of the Revised Wind Erosion Equation (RWEQ) model (Fryrear et al., 2000), a tool extensively tested to perform field-based predictions of soil loss due to wind erosion. RWEQ-GIS computes the soil loss potential on a daily basis for each 1000 m cell during the period between January 2001 and December 2010, by combining soil properties and daily data of rainfall, wind speed, evapotranspiration, soil moisture and crop canopy cover.

Total soil loss by erosion was calculated by summing soil loss by water and wind erosion, respectively.

Soil loss by water erosion, wind erosion, and total erosion, respectively, on land classified as annual crop production (see Section 2.4), was then averaged for each landscape. Thresholds for classification were applied based on Panagos et al. (2015; 2016) as specified in Table 1.

2.2.3. 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 composed of the following steps: (1) Distributed hydrological model setup and calibration; (2) Simulation of a long-term discharge time series and derivation of peak flows with selected return period; (3) Downscaling to 100 m spatial resolution and derivation of design flood hydrographs; and (4) Floodplain hydraulic simulations and merging of output flood depth maps (Alfieri et al., 2014). To indicate the degree to which individual landscapes are prone to recurring floods, the share of the total area in each landscape subject to 10-year flooding events was calculated for each landscape. Thresholds for classification were then applied as specified in Table 1.

2.2.4. Accumulated losses of soil organic carbon

Indicated by ``average SOC saturation capacity on land used for production of annual crops''. Data on SOC saturation capacity (expressed as the ratio of current SOC relative to the theoretical maximum potential) were taken from a published dataset with 250 m resolution, available at ESDAC. The data were created using a simulation platform that integrates the CENTURY agroecosystem model (Parton et al., 1988) with several Pan-European spatial and statistical databases (Lugato et al., 2014b) and simulates the changes in SOC over the period 2013–2100 by replacing current land use with alternative management practices (Lugato et al., 2014a). The data used for the purpose of this study represents the conversion of current land use to grassland, as this scenario resulted in the largest positive gain in SOC overall in Europe (Lugato et al., 2014a). For each landscape, the average SOC saturation capacity on land used for annual crop production was then calculated. Thresholds were defined using geostatistical properties to define five equal intervals between the minimum and maximum aggregated average SOC saturation capacity values, based on expert (i.e. model developer) recommendations (Table 1).

For this impact, high (10% higher threshold values) and low (10% lower threshold values) estimates were also defined, to enable a sensitivity test (see Section 2.7).

2.3. Mitigation potential of strategic perennialization

Perennialization in the form of wind breaks can increase yields for annual crops on land protected from wind, due to reduced crop damages (e.g., plant blasting, coverage of plants, uncovered roots and seeds), while also avoiding losses of organic matter and fine soil particles that can lead to decreased soil fertility. To be effective, windbreak cultivations need to be several meters high, hence preferably based on woody crops. For example, 50-meter wide willow plantations located 100 m apart can provide continuous sheltering in areas exposed to wind erosion and on sensitive soils, if half of the plantation width is harvested at a time (Börjesson, 1999a).

Perennial cultivations can be used as riparian buffer strips and filter zones reducing nutrient (and other agrochemical) emissions from arable land. Plantations designed and managed similarly as for windbreaks can be located along open waterways to continuously capture nutrients (Berndes et al., 2008; Styles et al., 2016; Ferrarini et al., 2017). Riparian buffer zones may consist of perennial grass cultivations and/or short-rotation woody plantations. Field trials have shown that N removal rates between herbaceous and woody crops, and between planted and spontaneous crops, are comparable (Ferrarini et al., 2017). A 20 m buffer with SRC and/or grass has been suggested to have 100% nitrate removal effectiveness (Ferrarini et al., 2017). However, several different designs have been suggested in the literature, from 50 m with SRC willow (Styles et al., 2016) to 5 m with grass (Ferrarini et al., 2017). On arable land with covered drainage systems, nutrient-rich drainage water can be collected in storage ponds and used for irrigation. Besides efficient nutrient retention and water purification, the irrigation can improve yield levels and reduce the need for commercial fertilizers (Börjesson and Berndes, 2006). Vegetation zones, or strips of perennial crop cultivations, can also be located in areas sensitive to rill erosion, particularly on fields with clayey and silty soils in hilly areas (Börjesson, 1999a). Prevention of water erosion requires continuous soil cover, which can make perennial grass cultivations preferable to short-rotation woody plantations. Similar types of vegetation zones can also be used for flood prevention (Berndes et al., 2008). Besides the onsite benefits of reduced soil losses, there are also offsite benefits, such as reduced sediment loading in reservoirs and irrigation channels, as well as reduced deterioration in the quality of river water due to the suspended load that accompanies flood waters formed mostly by runoff.

Independently of the type of perennial cultivation, replacement of annual crops with perennial crops normally leads to increased soil carbon sequestration (Whitaker et al., 2018). This is due to a combination of an increased input of organic matter to the soil and reduced soil tillage, leading to decreased decomposition of soil organic matter by microorganisms. Thus, this benefit will normally be provided in all situations where annual crops are replaced (Berndes et al., 2008). The extent may however vary geographically, due to local and regional climate conditions as well as the historical land use, e.g., the intensity in previous cultivation of annual crops (Berndes et al., 2012). This is also illustrated in the concept of SOC saturation capacity (Lugato et al., 2014a; 2014b), used as indicator for accumulated SOC losses in this study.

2.4. Annual crop dominance

The introduction of perennial crops for mitigating environmental impacts can only be effective in landscapes dominated by the production of annual crops, which has caused the environmental impacts by degrading the regulating ES supply. To estimate the effectiveness of perennialization, the *annual crop dominance*, i.e., the share of land in each landscape used for the production of annual crops compared with the total vegetated area, was calculated for each landscape.

The share of annual crops in each landscape was calculated using the CORINE 2012 100 m LULC dataset (Copernicus Land Monitoring Service, 2018). The CORINE raster was first reclassified from 47 to four land use classes, ``annual crops'', ``other agriculture'', ``other vegetation'' and ``unvegetated'' (Table 2). The number of 100 m cells was then calculated for each of the four land use classes within each landscape

Table 2
Reclassification of land use classes in CORINE 2012.

Aggregated land use class	CORINE land use class (GRID_CODE)
1: Annual crops	12, 13
2: Other agriculture	14–22
3: Other vegetation	10–11, 23–29, 32–33, 35–39, 49
4: Unvegetated	1–9, 30–31, 34, 40–44, 50
Null[a]	48

[a] Refers to cells classified as ``NODATA" in the original dataset.

unit. Finally, the share of annual crops of all vegetation was calculated in each landscape (annual crops / (annual crops + other agriculture + other vegetation)).

Thresholds for *annual crop dominance* classes were defined based on univariate statistics, as specified in Table 3. The distribution was skewed (mean: 0.33, median: 0.27, skewness: 0.62) so quantiles were used to define reasonable thresholds. Note that landscapes without annual crops were excluded in the computation of quantiles but still (naturally) classified as *very low* annual crop production dominance. This class therefore has significantly more observations than other classes.

2.5. Mitigation effectiveness of strategic perennialization

The annual crop dominance and the estimated degree of the five environmental impacts were combined to define four levels of expected effectiveness of perennialization, as illustrated in Table 4. This level was calculated for each environmental impact in each landscape.

2.6. Priority areas for strategic perennialization

Priority areas for beneficial LUC are conceptually referred to as landscape units where the environmental effects of perennialization are estimated to be particularly beneficial. In the modeling framework, priority areas are defined as landscapes where

1 one environmental impact could be mitigated with *very high* effectiveness, or
2 multiple impacts could be mitigated with either *high* or *very high* effectiveness

To identify the latter, the number of impacts for which perennialization was classified as having a *high* and *very high* expected effectiveness, respectively, were identified (see Section 2.5) and counted for each landscape.

2.7. Sensitivity analysis

Accumulated SOC losses had a very high influence in the identification of priority areas (see Results). To test how sensitive the identification of priority areas is to variations in threshold definitions, a high (thresholds increased with 10%) and low (thresholds decreased with 10%) estimate of accumulated SOC losses (Table 1) were used in the identification of priority areas.

3. Results

3.1. Effectiveness of strategic perennialization

The extent to which the assessed environmental impacts can be mitigated by perennialization depends on the degree of environmental impact in the landscape, and the dominance of annual crops relative to other vegetation. As summarized in Table 5 (see Table S1 and S2 for more information) and detailed below, the results indicate that there is a substantial potential for effective mitigation regarding all the assessed impacts.

The production of annual crops is an important determinant for accumulated loss in SOC. For other impacts, the spatial correlation is weaker, indicating that there are additional important biophysical factors influencing the degree of soil erosion, nitrogen emissions to water, and recurring floods. For example, nitrogen emissions to water can be very high in areas with high precipitation and/or intensive livestock production, even if the land is largely covered by perennials (see, e.g., Ireland in Fig. 1). The same can be seen for soil loss by water erosion which can be high in mountainous areas or on land with steep slopes, regardless of the land use. Soil loss by wind erosion, the least severe impact overall, is largely driven by wind exposure, hence mainly limited to coastal areas or higher altitudes, but also by structural deficits and topsoil texture. It can be observed that where several contributing parameters co-exist, the degree of environmental impact is particularly high.

3.1.1. Nitrogen emissions to water

Nitrogen emissions to water is classified as high to very high in 9% of all landscapes in EU28, containing 11% of the total area under annual crop production (Table 5). The majority of these landscapes are located in north-western Europe; most notably in Ireland, Western UK, Denmark, and the Netherlands (Fig. 1).

Mitigation of nitrogen emissions to water by strategic perennialization could be achieved with *high* or *very high* effectiveness in 4.4% of all landscapes, containing 12% of the total area under annual crop production (Table 5). As for the impact, the mitigation effectiveness is significant mainly in north-western Europe; primarily in large parts of the UK and Denmark, as well as parts of the Netherlands and Belgium, northern France, western Germany, the Po Valley in Italy and in the western parts of the Danube basin (Fig. 1).

3.1.2. Soil loss by erosion

Soil loss by water erosion is classified as high to very high in 12% of all landscapes in EU28, containing 12% of the total area under annual crop production (Table 5). The majority of these landscapes are located in southern Europe; most notably in large parts of Italy and parts of Spain, Romania, Slovakia, and southern Poland (Fig. 2).

Soil loss by wind erosion is a lesser concern, in general; classified as high to very high in 0.4% of all landscapes in EU28, containing 1% of the total area under annual crop production (Table 5). The majority of these landscapes are located in western UK, Denmark, the Netherlands and eastern Bulgaria (Fig. 2).

Total loss by wind and water erosion combined is classified as high

Table 3
Definitions of annual crop dominance classes and resulting number of landscapes, corresponding landscape area, and affected area under annual crop production.

Annual crop dominance	Percentile	% annual crops of total vegetated area within landscape	Landscapes #	Landscapes % of total #	Total area Thousand hectares	Total area % of total	Area with annual crop production Thousand hectares	Area with annual crop production % of total ha
Very low	0–15	≤ 3.38983	39 595	49%	138 980	33%	637	1%
Low	15–35	(3.38983,14.1245]	9 854	12%	60 626	14%	4 692	4%
Medium	35–65	(14.1245,41.8919]	14 780	18%	94 613	22%	24 163	22%
High	65–85	(41.8919,66.8304]	9 853	12%	73 444	17%	36 915	34%
Very high	85–100	> 66.8304	7 390	9%	57 869	14%	43 191	39%

Table 4
Expected effectiveness of perennialization in mitigating negative environmental impacts by enhancing corresponding ecosystem services. Colours indicate marginal (blue), low (purple), medium (light red), high, (orange), and very high (yellow) expected effectiveness. Colours are identical as in Figs. 1–5.

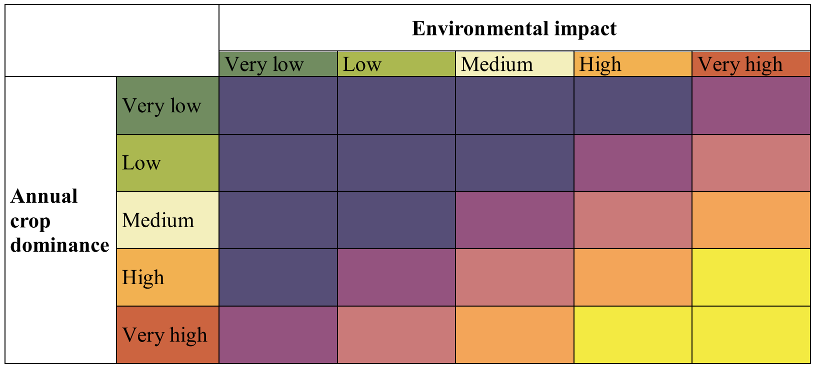

Table 5
Degree of environmental impacts and mitigation effectiveness of strategic perennialization in European landscapes. More information is available in Table S1 and S2.

		Degree of environmental impact			Effectiveness of strategic perennialization		
		% of total number of landscapes[a]	Area under annual crops Thousand hectares	% of total[a]	% of total number of landscapes[a]	Area under annual crop production Thousand hectares	% of total[a]
Nitrogen emissions to water	Very low / Marginal	60%	56 589	52%	77%	41 247	38%
	Low	22%	28 925	26%	14%	37 214	34%
	Medium	9%	12 865	12%	5%	16 960	15%
	High	4%	6 193	6%	4%	12 552	11%
	Very high	5%	5 025	5%	0,4%	1 626	1%
Water erosion	Very low / Marginal	40%	44	0%	65%	16 061	15%
	Low	30%	67 222	61%	15%	30 639	28%
	Medium	19%	29 928	27%	13%	44 921	41%
	High	8%	8 503	8%	6%	17 380	16%
	Very high	4%	3 901	4%	0,2%	596	0,5%
Wind erosion	Very low / Marginal	50%	2 873	3%	79%	29 382	27%
	Low	47%	99 288	91%	12%	36 256	33%
	Medium	2%	6 025	5%	8%	38 507	35%
	High	0.3%	1 196	1%	1%	5 266	5%
	Very high	0.1%	216	0%	0,04%	187	0,2%
Total erosion[b]	Very low / Marginal	40%	44	0%	64%	14 804	14%
	Low	26%	53 169	49%	15%	28 314	26%
	Medium	21%	39 735	36%	13%	37 797	34%
	High	9%	12 245	11%	8%	27 742	25%
	Very high	5%	4 404	4%	0,3%	941	0,9%
Recurring floods	Very low / Marginal	64%	51 303	47%	78%	42 470	39%
	Low	19%	33 500	31%	11%	31 579	29%
	Medium	6%	9 464	9%	5%	18 198	17%
	High	6%	9 121	8%	5%	13 373	12%
	Very high	5%	6 210	6%	1%	3 978	4%
Accumulated soil organic carbon losses	Very low / Marginal	44,0%	1 726	2%	58%	6 491	6%
	Low	5,4%	4 631	4%	12%	11 724	11%
	Medium	18,4%	26 149	24%	13%	22 458	20%
	High	29,9%	71 399	65%	16%	64 876	59%
	Very high	2,3%	5 692	5%	1%	4 049	4%

[a] Total percentage may differ from 100% due to rounding.
[b] Refers to the sum of soil loss by water and wind erosion. For example, a landscape may have ``high" water erosion and ``high" wind erosion resulting in either a ``high" or ``very high" total erosion, depending on the total amount of soil loss compared with the classification thresholds.

to very high in 14% of all landscapes in EU28, containing 15% of the total area under annual crop production (Table 5).

Mitigation of soil loss by either wind or water erosion by strategic perennialization could be achieved with *high* or *very high* effectiveness in just over 8% of all landscapes, containing about a quarter of the total area under annual crop production (Table 5). The mitigation effectiveness is significant in areas scattered all over Europe, but most notably in Eastern UK, Denmark, Spain, Italy, Romania, Bulgaria, and southern Poland (Fig. 3).

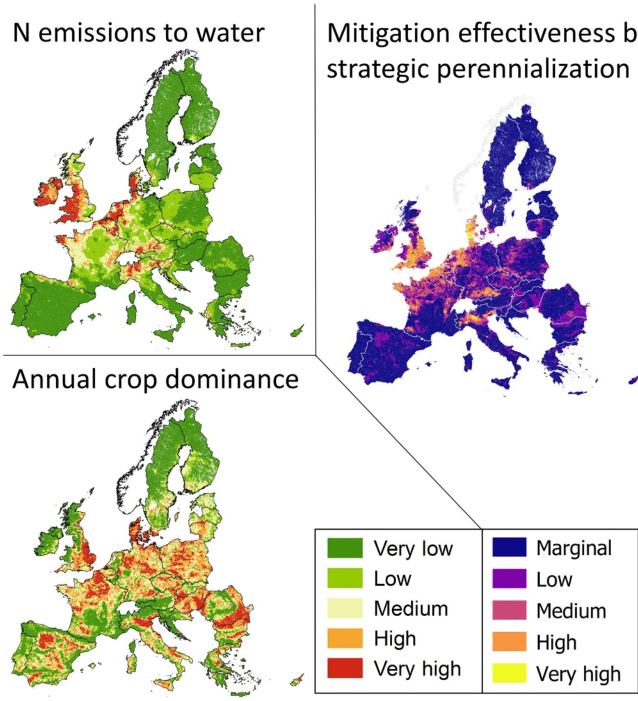

Fig. 1. Nitrogen emissions to water, annual crop dominance, and resulting mitigation effectiveness of strategic perennialization.

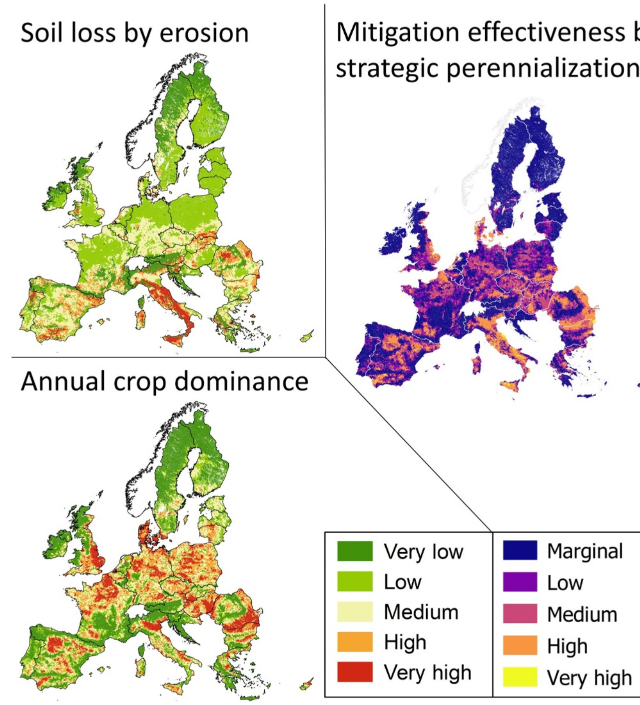

Fig. 3. Soil loss by erosion, annual crop dominance, and resulting mitigation effectiveness of strategic perennialization.

3.1.3. Recurring floods

Recurring floods is classified as high to very high in 11% of all landscapes in EU28, containing 14% of the total area under annual crop production (Table 5). These landscapes are primarily located along major rivers, such as the Danube, Po, Elbe, Oder, Vistula, and Rhône (Fig. 4).

Mitigation of recurring floods by strategic perennialization could be achieved with *high* or *very high* effectiveness in 6% of all landscapes, containing 16% of the total area under annual crop production (Table 5). The mitigation effectiveness is significant mainly in the Po Valley in Italy and along the Danube basin, but also in areas around other rivers throughout Europe (Fig. 4).

3.1.4. Accumulated loss of soil organic carbon

Accumulated losses of SOC is classified as high to very high in about a third of all landscapes in EU28, containing 70% of the total area under annual crop production (Table 5). These landscapes are scattered all over Europe, having a strong spatial correlation with the production of annual crops (Fig. 5).

Mitigation of SOC losses by strategic perennialization could be achieved with *high* or *very high* effectiveness in 17% of all landscapes, containing almost two thirds of the total area under annual crop production (Table 5). The mitigation effectiveness is significant in areas all over Europe; primarily in eastern UK, northern France, and large parts of Denmark, Italy, Spain, Germany, Poland, Lithuania, Czech Republic, Hungary, Romania and Bulgaria (Fig. 5).

3.2. Priority areas for strategic perennialization

The majority of annual crops cultivated in EU is located in landscapes where strategic perennialization can help mitigating different environmental impacts, in different ways and to different extents. Areas

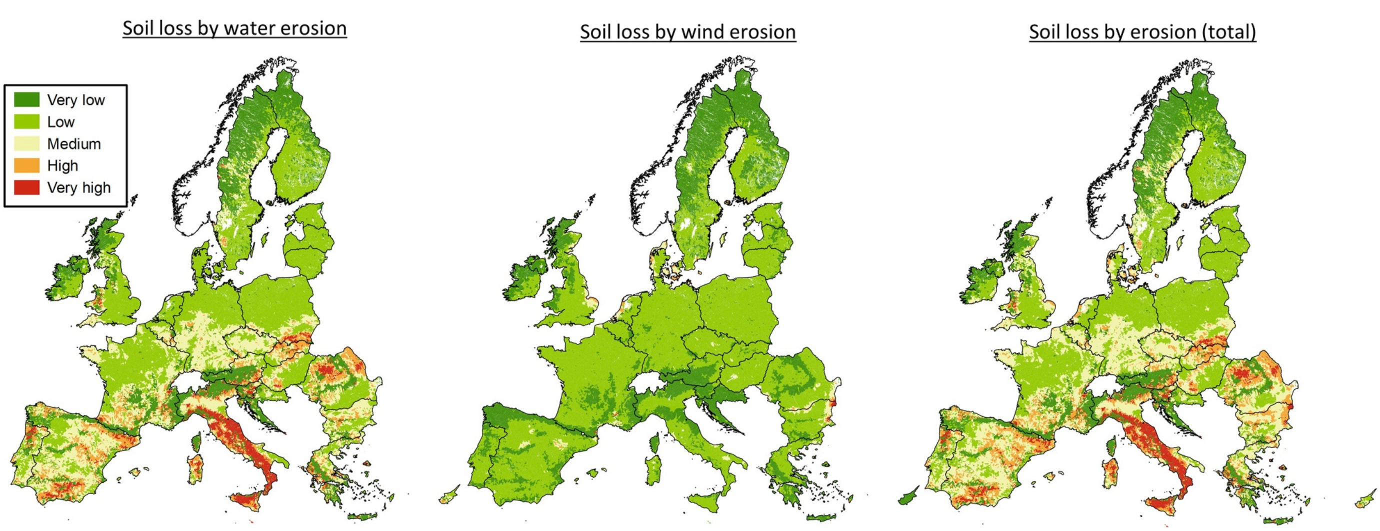

Fig. 2. Soil loss by water-, wind-, and total erosion, respectively.

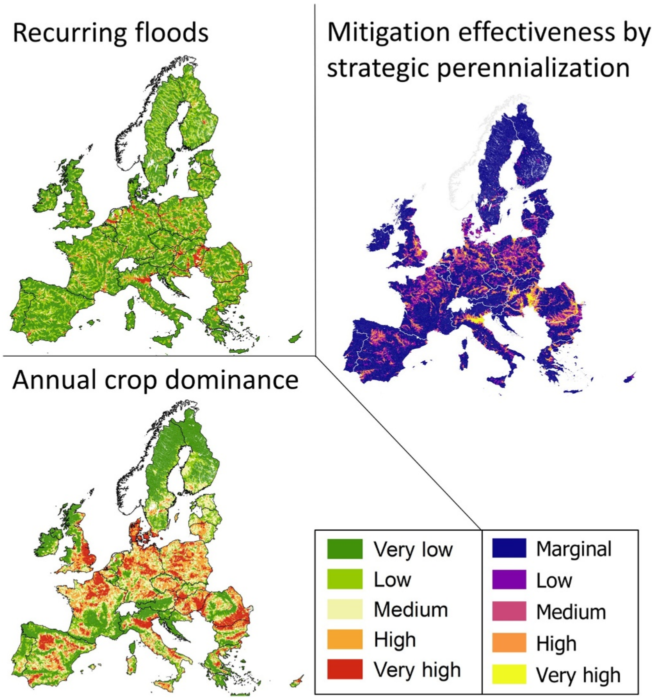

Fig. 4. Recurring floods, annual crop dominance, and resulting mitigation effectiveness of strategic perennialization.

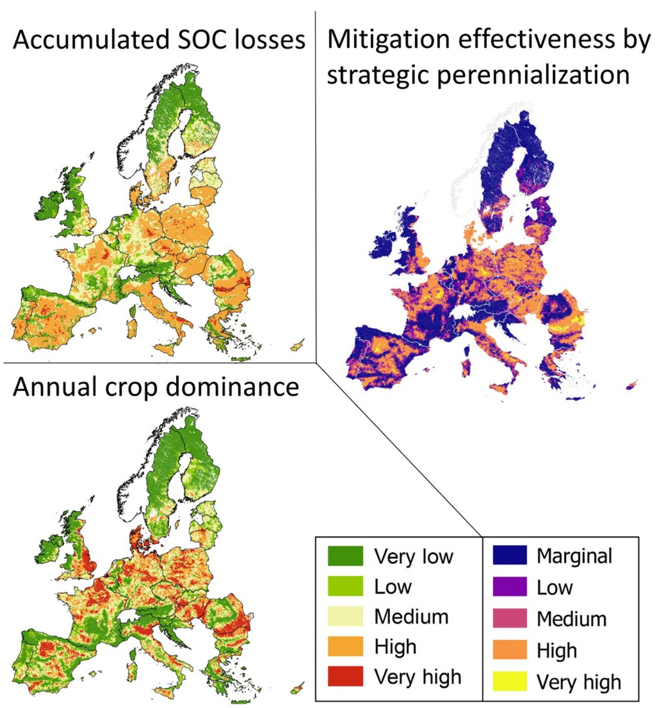

Fig. 5. Accumulated losses of soil organic carbon, annual crop dominance, and resulting mitigation effectiveness of strategic perennialization.

where perennialization can be particularly beneficial, from an environmental perspective, are here identified as *Priority areas* for beneficial LUC.

A total of 1764 landscapes, harboring 9% of total annual crop production in EU, can be considered priority areas, due to expected mitigation of a *single* environmental impact by perennialization with *very high* effectiveness (Table 6, see also Table S3 for more information).

Priority areas could also be defined as landscapes where *multiple* impacts can be mitigated with either *high* or *very high* effectiveness. Depending on the required number of impacts to be mitigated, such priority areas contain 1% (for four mitigated impacts), 9% (at least three impacts), or 37% (at least two impacts) of total annual crop production in EU, respectively (Table 6).

Combined, these two types of priority areas cover 15–60 million hectares, harboring 10–46% of total annual crop production in EU. These areas are scattered all over Europe, but there are notable "hot-spots" where priority areas are concentrated. This can be seen in, e.g., large parts of Denmark, western UK, The Po valley in Italy, and the Danube basin, but also in northern France, and several regions in, e.g., Spain, Germany, and Italy (Fig. 6).

3.3. Sensitivity analysis

Both the high and the low estimates of accumulated SOC losses had substantial effects on impact classification (Table S4). Effectiveness classification was less affected, which was expected as this is also influenced by the annual crop dominance (Table S4). There were, however, notable relative differences compared with the main analysis on the number of landscapes where perennialization was classified as having a *very high* mitigation effectiveness (Table S4). This can also be seen in the identification of priority areas defined as landscapes with *very high* expected mitigation effectiveness of a *single* environmental impact, where the low estimate resulted in a 25% decrease, and the high estimate resulted in a 51% increase, respectively, in the number of landscapes (Table 7). In total, the number of priority areas decreased with 13% in the low estimate and increased with 16% in the high estimate (Table 7).

Spatial patterns of impact and effectiveness classification, respectively (Fig. S1), as well as of priority areas (Figs. 6-7), in both the high and the low estimate are comparable with the main analysis. Priority "hot spots" are therefore very similar (Figs. 6-7).

4. Discussion

While the results indicate that large areas under annual crops could be subject to strategic perennialization, only parts of these areas would need to be converted to perennial systems. The area that need to be converted for achieving successful impact mitigation basically depends on the type and degree of the impact and what management system is implemented, which in turn can be influenced also by other factors, such as practicality in terms of planting and harvesting (determining, e.g., size of plantations) and local preferences concerning the landscape aesthetics (determining, e.g., selecting woody or herbaceous crops). It also depends on how to interpret "successful" impact mitigation. For example, to completely restore accumulated losses of SOC throughout a landscape, the entire cropland area in this landscape need to be converted to, e.g., grassland and maintained as such for a long period of time. If this is not desirable, a smaller share of the cropland area could instead be converted to enhance SOC at the landscape scale. Furthermore, the area of riparian buffers needed to mitigate N emissions to water depends on the width of the strip (5–50 m; see Section 2.3), as well as the total length of rivers in the landscape. It is thus difficult at this point to provide estimates of areas needed for strategic perennialization, and their corresponding impact mitigation effectiveness and biomass production. Preliminary calculations for riparian buffers however indicate that it could suffice to convert about 1% of the total cropland area in EU, to establish 20 m wide buffer strips in all landscapes where the effectiveness of strategic perennialization for mitigating nitrogen emissions to water is classified as high or very high.

While regional and national assessments can indicate areas where strategic perennialization could be environmentally beneficial, the actual effects of introducing perennials in agricultural landscapes depend on crop selection, management system, location in the landscape, and

Table 6
The total number of landscapes and areas under annual crops where strategic perennialization can mitigate different numbers of environmental impacts, with a high and/or very high effectiveness. Numbers in the coloured rows can be linked to identically coloured areas in Fig 6. See Table S3 for more information.

Effectiveness of perennialization	Number of impacts	% of total number of landscapes	Area under annual crop production	
			Thousand hectares	% of total
High	0	78%	33 814	31%
	1	13%	41 217	38%
	2	7%	27 140	25%
	3	2%	6 661	6%
	4	0,1%	765	0,7%
Very high	0	98%	99 266	91%
	1	2%	10 070	9%
	2 [1)]	0,1%	262	0,2%
	3	-	-	-
	4	-	-	-
High or very high [2)]	0	78%	32 055	29%
	1	12%	37 326	34%
	2	8%	30 151	28%
	3	2%	8 757	8%
	4	0,2%	1 309	1%

1. These landscapes are only visualized as part of the "high or very high" category with 2-4 impacts. Overlaps are specified in table notes 2-4.
2. Of which 47 have two "very high" and zero "high"
3. Of which 38 have two "very high" and one "high"
4. Of which 15 have two "very high" and two "high"

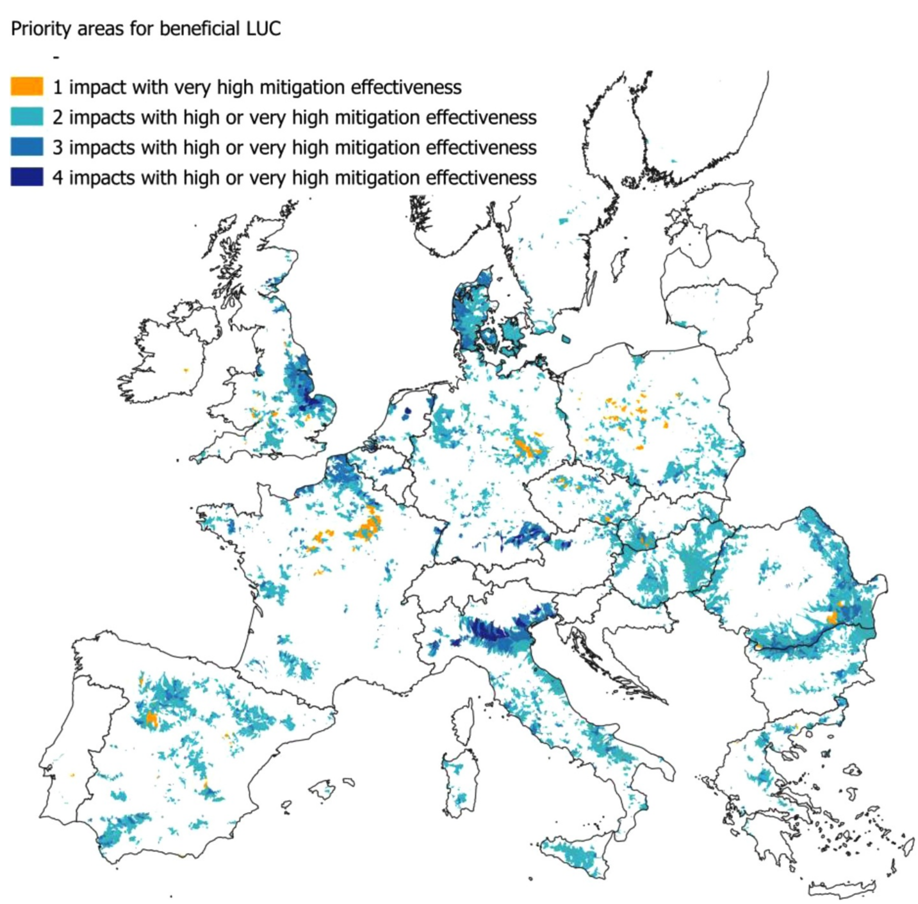

Fig. 6. Priority areas for beneficial LUC through strategic perennialization. In case a landscape appears in both the orange "*very high*" category and any of the blue "*high* to *very high*" categories (cf. Table 6), the latter is prioritized for visualization.

Table 7
Priority areas calculated as described in Section 3.2 using a high and low estimate of accumulated SOC losses, respectively.

		One impact mitigated with *very high* effectiveness	Two impacts mitigated with *high* or *very high* effectiveness	Three impacts mitigated with *high* or *very high* effectiveness	Four impacts mitigated with *high* or *very high* effectiveness	All priority areas
# landscapes	main analysis	1764	6492	1711	177	10,144
	low estimate	1341	5744	1565	161	8811
	high estimate	2671	7152	1792	192	11,807
% change	low estimate	−24%	−12%	−9%	−9%	−13%
	high estimate	51%	10%	5%	8%	16%

biotic and abiotic landscape characteristics. To fully understand—quantitatively as well as spatially—the effects of perennialization, high-resolution spatially explicit analysis within individual landscapes is required (Englund et al., 2017). One important characteristic to include in such assessments can be sub-landscape or sub-field variations in cropland productivity, as demonstrated by Ssegane et al. (2015). Targeting strategic perennialization to such land could provide environmental benefits with minimal impact on of total agricultural production. This could also inform farmers about alternative management systems for land with limited productivity, which can provide environmental benefits and possibly also increase total productivity at the farm level.

4.1. Model evaluation

The classification of environmental impacts and dominance of annual crops was supported by high-quality spatial models and datasets, calibrated and validated using empirical data, as summarized below. Further information is available in the original articles.

- *Nitrogen emissions to water* was estimated using the GREEN model, run specifically for the landscape dataset in this study. The model is calibrated using about 1400 measurement points of surface water quality and quantity, between 1985–2005. The Nash-Sutcliffe (1970) coefficient of efficiency of calibration was 92%, with yearly efficiencies ranging from 76% to 97%. The comparison between measured and estimated loads did not show any significant systematic or temporal deviations. For the 63 stations where complete time series were available, the correlation between the trends in measurements and in model estimates (computed as the slope of the linear interpolation) was 84%, indicating that the model is capturing rather well the observed temporal trends (Grizzetti et al., 2012).
- *Soil loss by water erosion* was estimated using a 100 m pan-European dataset derived from the RUSLE2015 model. The mean loss rates and spatial patterns are very close to national data reported in the EIONET-SOIL database for Germany, the Netherlands, Bulgaria, Poland and Denmark. It was found to be the most suitable modelling approach for estimating soil loss at the European scale, in terms of validation, usability, replicability, transparency, and parameterisation (Panagos et al., 2015).
- *Soil loss by wind erosion* was estimated using a 1000 m pan-European dataset from the GIS-RWEQ model. A cross-validation of the model showed that the predicted soil loss rates were generally in agreement with wind erosion sites reported in literature; 85 of 90 reported locations (94.4%) were classified by the model as being susceptible to erosion. Thereof, 23.3% of the literature sites fell into areas modelled as high erosion areas, whereas 48.9% fell into areas where slight to moderate erosion was predicted. The remaining 22.2% literature sites fell into areas classified as being very low to low erosive (Borrelli et al., 2017).
- *Recurring floods* was estimated using the first quantitative pan-European flood hazard assessment, representing the largest application of its kind at 100 m resolution. The map was evaluated against national/regional maps for three areas: the state of Saxony in Germany, the Thames, and the Severn River basin in the United Kingdom. Overall, the overlap between the pan-European and the national/regional maps ranges between 59% and 79%, depending on the region and the aggregation scale considered (Alfieri et al., 2014).
- *Accumulated SOC losses* was estimated using a 250 m pan-European dataset created using a simulation platform that integrates the CENTURY agroecosystem model with several pan-European spatial and statistical databases. Simulation values were validated against two independent empirical datasets, LUCAS and EIONET-SOIL. Simulated values showed a good agreement with measured values in

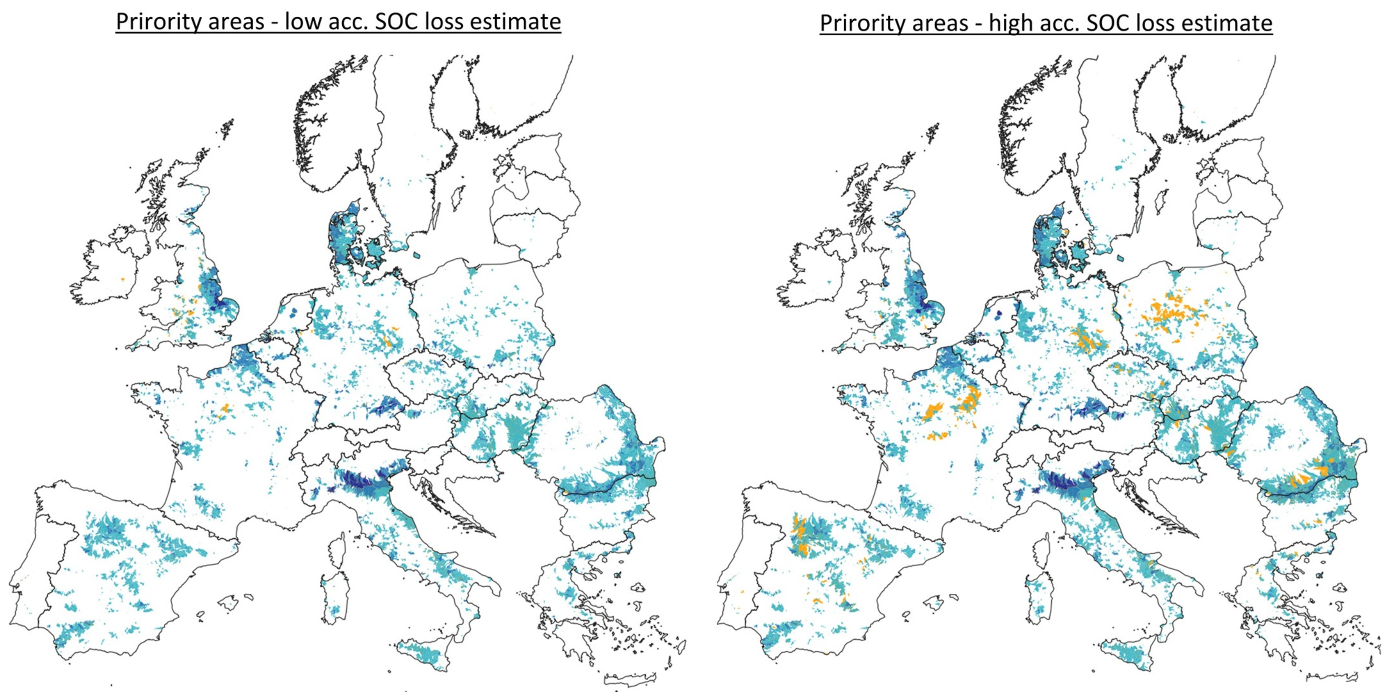

Fig. 7. Priority areas calculated using a high and low estimate of accumulated SOC losses, respectively. See Fig. 6 for legend and comparison with the main analysis.

all the aggregated land uses considered, with no particular bias (Lugato et al., 2014b).

- *Dominance of annual crops* was estimated using the 100 m CLC2012 land use dataset, produced within the CORINE Land Cover programme coordinated by the European Environment Agency. CORINE provides consistent information on land cover and land cover changes across Europe. CLC products are based on the photointerpretation of satellite images by the national teams of the participating countries. The resulting national land cover inventories are further integrated into a seamless land cover map of Europe (Copernicus Land Monitoring service, 2018).

The results are however sensitive to the threshold values used for the classification of negative impacts and annual crop dominance, as illustrated in Section 3.3. Impacts were classified based on advice from the model developers and providers of indicator datasets, except for the case of *recurring floods*, where the classification was based on arbitrarily defined thresholds. Thresholds for *annual crop dominance* classes were also arbitrarily defined based on univariate statistics. While results for individual landscapes are sensitive to threshold definitions, spatial patterns are generally not. The results presented here are therefore considered particularly useful for indicating relative differences between areas, and for identifying locations where perennialization can be particularly interesting from an environmental point of view. Such locations could later be subject of more detailed assessments, as discussed below.

4.1.1. Model extension and adaptation

The model presented here for estimating effectiveness of strategic perennialization can be further developed to, e.g. include additional environmental impact categories, and adapted to better suit, e.g., application at other geographical scales. As discussed above, additional analytical work is needed to provide estimates of areas for strategic perennialization, and corresponding impact mitigation effectiveness and perennial biomass production.

Applying the model at larger scales would be challenging, as sufficiently reliable data at high resolution is often lacking. It would require combining many different datasets and accepting large uncertainties. Applying the model at national scale can be done using national or regional datasets with higher precision, and thus produce more reliable results. Such results can also be more easily evaluated as the national context can be fully considered with the involvement of relevant stakeholders.

4.2. Policy considerations

Policies and regulations put in place to establish a societal transition towards the Paris targets will likely lead to an increased biomass demand for bioenergy and other bio-based products. Yet, despite that knowledge and practical experience from field trials and commercial applications have existed for several decades, perennialization activities of the type described in this study rarely takes place in EU. Studies commonly find significant socioeconomic values (Berndes et al., 2008; Börjesson, 1999b; Börjesson and Berndes, 2006), but the incentives for farmers to achieve such beneficial LUC have not been sufficiently strong. The Common Agricultural Policy (CAP) of the EU has historically not provided direct support for perennial plantations producing biomass feedstock for, e.g., energy purposes. Inadequate knowledge support, low biomass prices, and market uncertainty are other reasons behind slow development for production systems with perennial grasses and woody crops (Dimitriou et al., 2018; Dimitriou et al., 2011).

The effectiveness in promoting beneficial LUC may increase if policies and regulations seek synergies between climate change mitigation, energy security, and other societal goals, e.g., related to SDGs. Recent policy development is favorable in some areas. For example, the CAP currently requires that all arable areas exceeding 15 ha must set aside 5% of the area for ``ecologically beneficial elements" (Ecological Focus Areas, EFAs). The main purpose of EFAs is to enhance biodiversity, but also to provide other environmental benefits. EFAs can be in the form of, e.g., fallow land, terraces, landscape features, buffer strips, agroforestry, strips along forest edges, short rotation coppice with no use of fertilizers and/or plant protection products, catch crops, and nitrogen-fixing crops (European Parliament and the Council, 2013). The biomass produced on these areas is allowed to be used as feedstock for various purposes, including bioenergy. This may act as a driver for increased perennialization in agricultural landscapes, hence beneficial LUC.

Localization of EFAs in the landscape will be determined by biotic and abiotic landscape characteristics as well as stakeholder preferences. In some cases, EFAs may provide the highest environmental benefits by being scattered across the landscape, while in other cases it may be more beneficial to connect EFAs to provide green infrastructure, which would also simplify potential biomass harvesting. The approach presented in this article can be further developed to provide more detailed information on how to localize EFAs to meet different objectives in individual landscapes. Such information can facilitate landscape design processes where landowners, local decision makers, and other relevant stakeholders jointly develop strategies for beneficial LUC that reflect local conditions and preferences (Busch, 2017).

If the achievement of beneficial LUC causes losses in the production of agriculture commodities, the production of the same commodities will need to increase elsewhere, unless changes in demand and efficiency improvements along supply chains can fully buffer the losses. Effects of such indirect LUC (iLUC) need to be considered in relation to any measure that aim to reduce land use impacts, e.g., changes from conventional to organic agriculture, restrictions of fertilizer use to protect water, or lower stocking densities in animal agriculture.

In response to concerns that iLUC will cause large negative effects, various approaches to identify so-called low iLUC risk options have been developed (Peters et al., 2016). Options for achieving beneficial LUC through perennialization can provide opportunities to reduce land use impacts while achieving high biomass yields. The biomass can then be refined to multiple products, including biofuels and animal feed, hence substituting conventional (cultivated) feed and reducing grazing requirements (Egeskog et al., 2011; Larsen et al., 2017; Manevski et al., 2017, 2018; Solati et al., 2018; Sparovek et al., 2007). Such options can help maintain or increase agricultural production in a region while limiting environmental impacts, or reduce imports of agricultural commodities that are associated with negative impacts where they are produced. In other cases, when reduced food commodity production will be compensated by increased production elsewhere, this need not imply adverse environmental impacts; outcomes critically depend on the context where production increases, including governance of land use.

Beneficial LUC need not be premised on the requirement that the production of agriculture commodities in a region is not reduced. However, it remains important to consider possible iLUC impacts when evaluating how options for achieving beneficial LUC contribute to set policy objectives, such as GHG emissions reduction. These issues are further addressed in subsequent ongoing studies that quantifybiomass supply potentials and GHG mitigation associated with strategies for achieving beneficial LUC in EU.

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Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.gloenvcha.2019.101990.

References

Abad, C.R., Althaus, H.J., Berndes, G., Bolwig, S., Corbera, E., Creutzig, F., Ulloa, J.G., Geddes, A., Gregg, J.S., Haberl, H., Hanger, S., Harper, R.J., Hunsberger, C., Larsen, R.K., Lauk, C., Leitner, S., Lilliestam, J., Campen, H.L., Muys, B., Nordborg, M., Ölund, M., Orlowsky, B., Popp, A., Pereira, J.P., Reinhard, J., Scheiffle, L., Smith, P., 2017. Bioenergy production and sustainable development: science base for policymaking remains limited. GCB Bioenergy 9, 541–556. https://doi.org/10.1111/gcbb.12338.

Alfieri, L., Salamon, P., Bianchi, A., Neal, J., Bates, P., Feyen, L., 2014. Advances in pan-European flood hazard mapping. Hydrol. Process. 28, 4067–4077. https://doi.org/10.1002/hyp.9947.

Alston, J.M., Pardey, P.-.G., 2014. Agriculture in the global economy. J. Econ. Perspect. 28 (1), 121–146.

Asbjornsen, H., Hernandez-Santana, V., Liebman, M., Bayala, J., Chen, J., Helmers, M., Ong, C.K., Schulte, L.A., 2014. Targeting perennial vegetation in agricultural landscapes for enhancing ecosystem services. Renewable Agric. Food Syst. 29, 101–125. https://doi.org/10.1017/S1742170512000385.

Berndes, G., Ahlgren, S., Börjesson, P., Cowie, A.L., 2012. Bioenergy and land use change-state of the art. WENE 2, 282–303. https://doi.org/10.1002/wene.41.

Berndes, G., Börjesson, P., Ostwald, M., Palm, M., 2008. Multifunctional biomass production systems -an overview with presentation of specific applications in India and Sweden. Biofuels Bioprod. Bioref. 2, 16–25. https://doi.org/10.1002/bbb.52.

Berndes, G., Fredrikson, F., Börjesson, P., 2004. Cadmium accumulation and Salix-based phytoextraction on arable land in Sweden. Agric. Ecosyst. Environ. 103, 207–223. https://doi.org/10.1016/j.agee.2003.09.013.

Berndes, G., Hoogwijk, M., van den Broek, R., 2003. The contribution of biomass in the future global energy supply: a review of 17 studies. Biomass Bioenergy 25, 1–28. https://doi.org/10.1016/S0961-9534(02)00185-X.

Borrelli, P., Lugato, E., Montanarella, L., Panagos, P., 2017. A new assessment of soil loss due to wind erosion in European agricultural soils using a quantitative spatially distributed modelling approach. Land Degrad. Dev. 28, 335–344. https://doi.org/10.1002/ldr.2588.

Börjesson, P., 1999a. Environmental effects of energy crop cultivation in Sweden - I: identification and quantification. Biomass Bioenergy 16, 137–154. https://doi.org/10.1016/S0961-9534(98)00080-4.

Börjesson, P., 1999b. Environmental effects of energy crop cultivation in Sweden - II: economic valuation. Biomass Bioenergy 16, 155–170. https://doi.org/10.1016/S0961-9534(98)00081-6.

Börjesson, P., Berndes, G., 2006. The prospects for willow plantations for wastewater treatment in Sweden. Biomass Bioenergy 30, 428–438. https://doi.org/10.1016/j.biombioe.2005.11.018.

Burel, F., Baudry, J., 2003. Landscape Ecology. Science Publishers.

Busch, G., 2017. A spatial explicit scenario method to support participative regional land-use decisions regarding economic and ecological options of short rotation coppice (SRC) for renewable energy production on arable land: case study application for the Göttingen district, Germany. Energy Sustain. Soc. 7 (2). https://doi.org/10.1186/s13705-017-0105-4.

Cacho, J.F., Negri, M.C., Zumpf, C.R., Campbell, P., 2017. Introducing perennial biomass crops into agricultural landscapes to address water quality challenges and provide other environmental services. Wiley Interdiscip. Rev.: Energy Environ. 7, e275.

Christen, B., Dalgaard, T., 2013. Buffers for biomass production in temperate European agriculture: a review and synthesis on function, ecosystem services and implementation. Biomass Bioenergy 55, 53–67. https://doi.org/10.1016/j.biombioe.2012.09.053.

Christian, D.P., Niemi, G.J., Hanowski, J.M., Collins, P., 1994. Perspectives on biomass energy tree plantations and changes in habitat for biological organisms. Biomass Bioenergy 6, 31–39. https://doi.org/10.1016/0961-9534(94)90082-5.

Clarke, L., Jiang, K., Akimoto, K., Babiker, M., Blanford, G., Fisher-Vanden, K., Hourcade, J.C., Krey, V., Kriegler, E., Löschel, A., McCollum, D., Paltsev, S., Rose, S., Shukla, P.R., Tavoni, M., van der Zwaan, B., Van Vuuren, D.P., 2014. Assessing transformation pathways. In: Edenhofer, O., Pichs-Madruga, R., Sokona, Y., Farahani, E., Kadner, S., Seyboth, K., Adler, A., Baum, I., Brunner, S., Eickemeier, P., Kriemann, B., Savolainen, J., Schlömer, S., von Stechow, C., Zwickel, T., Minx, J.C. (Eds.), Climate Change Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel On Climate Change. Cambridge University Press, Cambridge, UK and New York, NY, USA, pp. 413–510.

Copernicus Land Monitoring Service, 2018. CLC 2012. Available at:https://land.copernicus.eu/pan-european/corine-land-cover/clc-2012.

Creutzig, F., Ravindranath, N.H., Berndes, G., Bolwig, S., Bright, R., Cherubini, F., Chum, H., Corbera, E., Delucchi, M., Faaij, A., Fargione, J., Haberl, H., Heath, G., Lucon, O., Plevin, R., Popp, A., Abad, C.R., Rose, S., Smith, P., Stromman, A., Suh, S., Masera, O., 2015. Bioenergy and climate change mitigation: an assessment. GCB Bioenergy 7, 916–944. https://doi.org/10.1111/gcbb.12205.

D'Amato, D., Droste, N., Allen, B., Kettunen, M., Lähtinen, K., Korhonen, J., Leskinen, P., Matthies, B., Toppinen, A., 2017. Green, circular, bio economy: a comparative analysis of sustainability avenues. J. Clean. Prod. https://doi.org/10.1016/J.JCLEPRO.2017.09.053.

Dauber, J., Miyake, S., 2016. To integrate or to segregate food crop and energy crop cultivation at the landscape scale? Perspectives on biodiversity conservation in agriculture in Europe. Energy Sustain. Soc. 6 (1), 6 201625.

Dimitriou, I., Rosenqvist, H., Berndes, G., 2011. Slow expansion and low yields of willow short rotation coppice in Sweden; implications for future strategies. Biomass Bioenergy 35 (11), 4613–4618.

Dimitriou, I., Berndes, G., Englund, O., Brown, M., Busch, G., Dale, V., Devlin, G., English, B., Goss, K., Jackson, S., Kline, K.L., McDonnell, K., McGrath, J., Mola-Yudego, B., Murphy, F., Negri, M.C., Parish, E.S., Ssegane, H., Tyler, D., 2018. Lignocellulosic Crops in Agricultural Landscapes: Production Systems for Biomass and other Environmental Benefits – Examples, Incentives, and Barriers. IEA Bioenergy Task 43 report TR2018:05Available at: http://task43.ieabioenergy.com/wp-content/

uploads/2018/12/TR2018-05.pdf.
Egeskog, A., Berndes, G., Freitas, F., Gustafsson, S., Sparovek, G., 2011. Integrating bioenergy and food production—A case study of combined ethanol and dairy production in Pontal, Brazil. Energy Sustainable Dev. 15, 8–16. https://doi.org/10.1016/j.esd.2011.01.005.
Englund, O., Berndes, G., Cederberg, C., 2017. How to analyse ecosystem services in landscapes—A systematic review. Ecol. Indic. 73, 492–504. https://doi.org/10.1016/j.ecolind.2016.10.009.
European Environment Agency, 2012. European Catchments and Rivers Network System (Ecrins). European Environment Agency (EEA), Copenhagen.
European Parliament and the Council, 2013. Regulation (EU) No 1307/2013 of the European Parliament and of the Council.
FAO, 2014. Contribution of the forestry sector to national economies, 1990-2011. In: Lebedys, A., Li, Y. (Eds.), Forest Finance Working Paper FSFM/ACC/09. Rome. FAO.
Ferrarini, A., Fornasier, F., Serra, P., Ferrari, F., Trevisan, M., Amaducci, S., 2017. Impacts of willow and miscanthus bioenergy buffers on biogeochemical N removal processes along the soil-groundwater continuum. GCB Bioenergy 9 (1), 246–261. https://doi.org/10.1111/gcbb.12340.
Fryrear, D.W., Bilbro, J.D., Saleh, A., Schomberg, H., Stout, J.E., Zobeck, T.M., 2000. RWEQ: improved wind erosion technology. J. Soil Water Conserv. 55 (2), 183–189.
Göransson, G., 1994. Bird fauna of cultivated energy shrub forests at different heights. Biomass Bioenergy 6, 49–52. https://doi.org/10.1016/0961-9534(94)90084-1.
Grigal, D.F., Berguson, W.E., 1998. Soil carbon changes associated with short-rotation systems. Biomass Bioenergy 14, 371–377. https://doi.org/10.1016/S0961-9534(97)10073-3.
Grizzetti, B., Bouraoui, F., Aloe, A., 2012. Changes of nitrogen and phosphorus loads to European seas. Global Change Biol. 18, 769–782. https://doi.org/10.1111/j.1365-2486.2011.02576.x.
Gustafsson, L., 1987. Plant conservation aspects of energy forestry - a new type of land-use in sweden. For. Ecol. Manage. 21, 141–161. https://doi.org/10.1016/0378-1127(87)90078-8.
Holland, R.A., Eigenbrod, F., Muggeridge, A., Brown, G., Clarke, D., Taylor, G., 2015. A synthesis of the ecosystem services impact of second generation bioenergy crop production. Renewable Sustainable Energy Rev. 46, 30–40. https://doi.org/10.1016/j.rser.2015.02.003.
IPCC, 2018. Global Warming of 1.5 °C.
IPCC, 2019. IPCC Special Report on Climate Change, Desertification, Land Degradation, Sustainable Land Management, Food Security, and Greenhouse gas fluxes in Terrestrial Ecosystems. Summary for Policymakers, Approved Draft 07 August 2019Available at: https://www.ipcc.ch/srccl-report-download-page/.
Kline, K.L., Dale, V.H., 2008. Biofuels: effects on land and fire. Science 321 (5886), 199. https://doi.org/10.1126/science.321.5886.199.
Kort, J., Collins, M., Ditsch, D., 1998. A review of soil erosion potential associated with biomass crops. Biomass Bioenergy 14, 351–359. https://doi.org/10.1016/S0961-9534(97)10071-X.
Lacoste, M., Minasny, B., McBratney, A., Michot, D., Viaud, V., Walter, C., 2014. High resolution 3D mapping of soil organic carbon in a heterogeneous agricultural landscape. Geoderma 213, 296–311. https://doi.org/10.1016/j.geoderma.2013.07.002.
Larsen, S., Bentsen, N.S., Dalgaard, T., Jørgensen, U., Olesen, J.E., Felby, C., 2017. Possibilities for near-term bioenergy production and GHG-mitigation through sustainable intensification of agriculture and forestry in Denmark. Environ. Res. Lett. 12, 114032. https://doi.org/10.1088/1748-9326/aa9001.
Leemans, R., van Amstel, A., Battjes, C., Kreileman, E., Toet, S., 1996. The land cover and carbon cycle consequences of large-scale utilizations of biomass as an energy source. Global Environ. Change 6, 335–357. https://doi.org/10.1016/S0959-3780(96)00028-3.
Lugato, E., Bampa, F., Panagos, P., Montanarella, L., Jones, A., 2014a. Potential carbon sequestration of European arable soils estimated by modelling a comprehensive set of management practices. Global Change Biol. 20, 3557–3567. https://doi.org/10.1111/gcb.12551.
Lugato, E., Panagos, P., Bampa, F., Jones, A., Montanarella, L., 2014b. A new baseline of organic carbon stock in European agricultural soils using a modelling approach. Global Change Biol. 20, 313–326. https://doi.org/10.1111/gcb.12292.
Manevski, K., Lærke, P.E., Jiao, X., Santhome, S., Jørgensen, U., 2017. Biomass productivity and radiation utilisation of innovative cropping systems for biorefinery. Agric. For. Meteorol. 233, 250–264. https://doi.org/10.1016/j.agrformet.2016.11.245.
Manevski, K., Lærke, P.E., Olesen, J.E., Jørgensen, U., 2018. Nitrogen balances of innovative cropping systems for feedstock production to future biorefineries. Sci. Total Environ. 633, 372–390. https://doi.org/10.1016/j.scitotenv.2018.03.155.
Milner, S., Holland, R.A., Lovett, A., Sunnenberg, G., Hastings, A., Smith, P., Wang, S., Taylor, G., 2016. Potential impacts on ecosystem services of land use transitions to second-generation bioenergy crops in GB. GCB Bioenergy 8, 317–333. https://doi.org/10.1111/gcbb.12263.
Nash, J.E., Sutcliffe, J.V., 1970. River flow forecasting through conceptual models: part 1. A discussion of principles. J. Hydrol. 10 (3), 282–290.
Panagos, P., Borrelli, P., Poesen, J., Ballabio, C., Lugato, E., Meusburger, K., Montanarella, L., Alewell, C., 2015. The new assessment of soil loss by water erosion in Europe. Environ. Sci. Policy 54, 438–447. https://doi.org/10.1016/j.envsci.2015.08.012.
Panagos, P., Imeson, A., Meusburger, K., Borrelli, P., Poesen, J., Alewell, C., 2016. Soil conservation in Europe: wish or reality? Land Degrad. Dev. 27, 1547–1551. https://doi.org/10.1002/ldr.2538.
Parton, W.J., Stewart, J.W.B., Cole, C.V., 1988. Dynamics of C, N, P, and S in grassland soils: A model. Biogeochemistry 5, 109–131.
Perttu, K.L., Kowalik, P.J., 1997. Salix vegetation filters for purification of waters and soils. Biomass Bioenergy 12, 9–19. https://doi.org/10.1016/S0961-9534.
Peters, D., Spöttle, M., Hähl, T., Kuhner, A.K., Cuijpers, M., Stomph, T.J., van der Werf, W., Grass, M., 2016. Methodologies for the Identification and Certification of Low ILUC Risk Biofuels. European Commission.
Power, A.G., 2010. Ecosystem services and agriculture: tradeoffs and synergies. Ecosystem services and agriculture: tradeoffs and synergies. Phil. Trans. R. Soc. B 365. https://doi.org/10.1098/rstb.2010.0143.
Rijtema, P.E., DeVries, W., 1994. Differences in precipitation excess and nitrogen leaching from agricultural lands and forest plantations. Biomass Bioenergy 6, 103–113. https://doi.org/10.1016/0961-9534(94)90089-2.
Rockström, J., et al., 2009. Planetary boundaries: exploring the safe operating space for humanity. Ecol. Soc. 14 (2), 32.
Scarlat, N., Dallemand, J.-F., Monforti-Ferrario, F., Nita, V., 2015. The role of biomass and bioenergy in a future bioeconomy: policies and facts. Environ. Dev. 15, 3–34.
Searchinger, T., Heimlich, R., Houghton, R.A., Dong, F.X., Elobeid, A., Fabiosa, J., Tokgoz, S., Hayes, D., Yu, T.H., 2008. Use of US croplands for biofuels increases greenhouse gases through emissions from land-use change. Science 319, 1238–1240.
Slade, R., Bauen, A., Gross, R., 2014. Global bioenergy resources. Nat. Clim. Change 4, 99–105. https://doi.org/10.1038/nclimate2097.
Smith, P., Haberl, H., Popp, A., Erb, K.-.H., Lauk, C., Harper, R., Tubiello, F.N., de Siqueira Pinto, A., Jafari, M., Sohi, S., Masera, O., Böttcher, H., Berndes, G., Bustamante, M., Ahammad, H., Clark, H., Dong, H., Elsiddig, E.A., Mbow, C., Ravindranath, N.H., Rice, C.W., Robledo Abad, C., Romanovskaya, A., Sperling, F., Herrero, M., House, J.I., Rose, S., 2013. How much land-based greenhouse gas mitigation can be achieved without compromising food security and environmental goals? Global Change Biol. 19, 2285–2302. https://doi.org/10.1111/gcb.12160.
Smith, P., Porter, J.R., 2018. Bioenergy in the IPCC assessments. GCB Bioenergy 10, 428–431. https://doi.org/10.1111/gcbb.12514.
Solati, Z., Manevski, K., Jørgensen, U., Labouriau, R., Shahbazi, S., Lærke, P.E., 2018. Crude protein yield and theoretical extractable true protein of potential biorefinery feedstocks. Ind. Crops Prod. 115, 214–226. https://doi.org/10.1016/j.indcrop.2018.02.010.
Sparovek, G., Berndes, G., Egeskog, A., de Freitas, F.L.M., Gustafsson, S., Hansson, J., 2007. Sugarcane ethanol production in Brazil: an expansion model sensitive to socioeconomic and environmental concerns. Biofuels Bioprod. Bioref. 1, 270–282. https://doi.org/10.1002/bbb.31.
Ssegane, H., Negri, M.C., Quinn, J., Urgun-Demirtas, M, 2015. Multifunctional landscapes: site characterization and field-scale design to incorporate biomass production into an agricultural system. Biomass-Bioenergy 80, 179–190. https://doi.org/10.1016/j.biombioe.2015.04.012.
Ssegane, H., Negri, M.C., 2016. An integrated landscape designed for commodity and bioenergy crops for a tile-drained agricultural watershed. J. Environ. Qual. 45, 1588–1596. https://doi.org/10.2134/jeq2015.10.0518.
Styles, D., Börjesson, P., D'Hertefeldt, T., Birkhofer, K., Dauber, J., Adams, P., Patil, S., Pagella, T., Pettersson, L.B., Peck, P., Vaneeckhaute, C., Rosenqvist, H., 2016. Climate regulation, energy provisioning and water purification: quantifying ecosystem service delivery of bioenergy willow grown on riparian buffer zones using life cycle assessment. Ambio 45, 872–884. https://doi.org/10.1007/s13280-016-0790-9.
Turner, M.G., 1989. Landscape ecology - the effect of pattern on process. Annu. Rev. Ecol. Syst. 20, 171–197. https://doi.org/10.1146/annurev.es.20.110189.001131.
Whitaker, J., Field, J.L., Bernacchi, C.J., Cerri, C.E.P., Ceulemans, R., Davies, C.A., DeLucia, E.H., Donnison, I.S., McCalmont, J.P., Paustian, K., Rowe, R.L., Smith, P., Thornley, P., McNamara, N.P., 2018. Consensus, uncertainties and challenges for perennial bioenergy crops and land use. GCB Bioenergy 10, 150–164. https://doi.org/10.1111/gcbb.12488.
Zumpf, C., Ssegane, H., Negri, M.C., Campbell, P., Cacho, J., 2017. Yield and water quality impacts of field-scale integration of willow into a continuous corn rotation system. J. Environ. Qual. 46, 811–818. https://doi.org/10.2134/jeq2017.02.0082.