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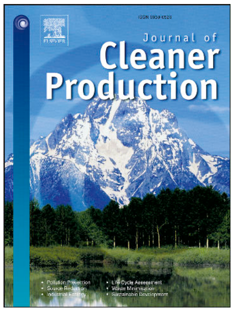

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1. Introduction

Increased agricultural production per unit of land area and per unit labour was achieved in many parts of Western Europe during the late twentieth century by using improved genetic material, increased inputs of irrigation, fertilisers, and agrochemicals, and increased use of large-scale specialized machinery that provided economies of scale (Burgess and Morris, 2009). However, these production and efficiency gains have often been achieved at the expense of the environment including water pollution (leaching and runoff of nitrogen, phosphorus and pesticides), soil degradation (e.g. erosion, compaction and loss of soil organic matter), loss of biodiversity, and increased greenhouse gas (GHG) emissions such as CO_2, CH_4 and N_2O (Garnett et al., 2013; Brown et al., 2018).

The governments of European countries such as the UK, Germany and France have indicated that they aim to achieve net zero emissions of greenhouse gases by 2050 (UK Government, 2019; European Parliament, 2019). One of the means to achieve net zero emissions is to increase carbon sequestration by promoting the growth of trees on farms using agroforestry (Hernández-Morcillo et al., 2018; Blazer et al., 2018; Kay et al., 2019a). Agroforestry has been defined as the deliberate integration of woody vegetation (trees and shrubs) with crop and/or animal systems to benefit from the resulting ecological and economic interactions (Burgess and Rosati, 2018). A systematic study has shown that the area of agroforestry in the Europe Union is equivalent to about 8.8% of the utilised agricultural land (den Herder et al., 2017). In addition to sequestering carbon, agroforestry can improve water quality by minimising the leaching of nutrients and pesticides (Jose, 2009; Nair, 2011, Jørgensen et al., 2018). Other benefits of agroforestry systems include an increase in biodiversity relative to monoculture crop or forest systems (Torralba et al., 2016; Blazer et al., 2018), improved soil conservation relative to monoculture arable systems (Herzog, 2000), and the socio-cultural value of historic land uses (Wolpert et al., 2020)

Agroforestry is a historic land use in Europe, with systems dating back centuries or millenia in some areas (Mosquera-Losada et al., 2012). In such areas, agroforestry landscapes are recognised for being part of a rural cultural heritage (Torralba et al., 2018b) and for their intrinsic aesthetic values (Torralba et al., 2018a). Recreation, aesthetic values and social interactions are also important ecosystem services of multifunctional landscapes (van Zanten et al., 2016; Oteros-Rozas et al., 2018; Fagerholm et al., 2019). In a meta-analysis for European agricultural landscapes, van Zanten et al., (2014) report the high values placed by people on mosaic land covers. Landscape diversity, naturalness, uniqueness and accessibility are also among the important features that people value in rural recreation areas (Boll et al., 2014; Paracchini et al., 2014).

The maintenance and uptake of agroforestry practices in Europe is determined by a range of socio-cultural, political, technical and natural factors (Wolpert et al., 2020), but the profitability of such systems relative to tree-only and arable-only systems can be pivotal to many farmers (García de Jalón et al., 2018a). Tree-only systems may be fruit orchards or woodlands; arable-only systems include rotations of annual crops sometimes including grass. One way in which governments can promote agroforestry is to provide economic incentives. In the European Union, such incentives are regulated by the Common Agricultural Policy (CAP) which comprises two pillars. Pillar I designates which farmed areas can receive basic farm payments and Pillar II describes the economic support for a range of rural development and agri-environmental measures (Mosquera-Losada et al., 2018).

In Europe, Graves et al., (2007) completed multi-site comparisons of the financial performance of agroforestry relative to arable and tree-only systems, but they did not quantify the environmental impacts. There have also been financial and economic analyses (including environmental externalities) for agroforestry relative to arable and tree-only systems for a single site (García de Jalón et al., 2018b) or a single area (Ovando et al., 2017). Kay et al., (2019b) compared the financial and economic impacts of agroforestry relative to arable systems but, with the exception of one site, did not consider tree-only systems. Hence, the objective of this article is to show the results of a study which developed an approach to compare the economic benefits of arable, agroforestry and tree-only systems at a plot-scale (1 ha) for three European case study sites (United Kingdom, Spain and Switzerland), considering five major environmental externalities: carbon dioxide emissions, carbon sequestration, the loss of nitrogen, the loss of phosphorus, and soil erosion. In addition, the paper determines the environmental externality values (€ unit^{-1}) at which agroforestry and tree-only systems achieve financial parity with arable cropping. Although the results are derived from three European case studies, the approach should be applicable to other areas.

2. Materials and Methods

2.1 Case study sites and selection of land use systems

We compared the profitability and economic benefits of arable, agroforestry and tree-only systems using three contrasting case studies from the UK, Spain, and Switzerland. The selected systems (Table 1) were identified as typical enterprises that are or could be used at each site. The first case study focused on Bedfordshire in lowland England. Here we compared a four-year arable crop rotation, a poplar agroforestry system with an understorey arable crop for 14 years and then put to grass fallow for the remaining 16 years, and a plantation of poplar also planted in year 1 and harvested in year 30. The second case study was located in the dehesa in Extremadura in Spain and compared an oat and grass rotation, a holm oak dehesa silvopastoral system, and a holm oak woodland starting from tree planting. For the third case study in Schwarzbubenland in north-west Switzerland, the arable system was a four-year crop rotation of oilseed rape, wheat, grass and wheat; the tree-only system was a cherry tree plantation for timber production, and the agroforestry system was a grassland with cherry trees used for fruit production (Table 1). The next four stages of the method were to: (i) simulate the biophysical growth of trees and crops, (ii) assess the financial performance from a farmer's perspective, (iii) quantify five environmental externalities, and (iv) express the environmental externalities in monetary terms. The last step was to determine the price (€ unit^{-1}) of the studied externalities which enabled the agroforestry or tree-only system to break-even with the arable system.

Table 1. Arable, agroforestry, and tree-only systems were compared at each of three case study sites

	Bedfordshire United Kingdom	Extremadura Spain	Schwarzbubenland, Switzerland
Length of rotation (years)	30	60	60
Arable system	Wheat Wheat Barley Oilseed	Oat Grass	Oilseed Wheat Grass Wheat
Agroforestry system	Same arable rotation[a] and *Populus* spp. 100 trees ha^{-1}	Grass, cows and *Quercus ilex* L. 50 trees ha^{-1} [b]	Grass, cows and *Prunus avium* 80 trees ha^{-1}
Tree-only system and thinning regime	*Populus* spp. 156 trees ha^{-1}	*Quercus ilex* L. [b] Year 0-27: 600 trees ha^{-1} Year 28-44: 425[c] Year 45-60: 250	*Prunus avium* Year 0-13: 816 trees ha^{-1} Year 14-29: 458[c] Year 30-60: 100

[a] Arable cropping did not continue after year 14; [b] An uneven-aged system with *Quercus ilex* L. trees was assumed; [c] Tree-only thinning regime (Year: residual trees)

2.2 Biophysical simulation

Simulated daily temperature, solar radiation and rainfall data for each site were obtained using the CliPick tool (Palma 2017). The annual rainfall and annual temperatures at the Bedfordshire site (59 m a.s.l.) ranged from 410 to 867 mm and from 9.1 to 11.3°C respectively. The Extremadura site is a gently sloping area 300-500 m a.s.l. The climate is dominated by very hot dry summers and wet winters with an annual rainfall of 500-600 mm and a mean annual temperature of 14.0-17.0°C. The site in Schwarzbubenland is the highest site (556 m a.s.l.) with a mean rainfall and temperature of 900 mm and 5.5°C respectively.

For each site, tree and arable crop growth and yields were predicted using the Yield-SAFE biophysical model (van der Werf et al., 2007), which was updated to include the Rothamsted Carbon model (RothC) to calculate changes in soil organic carbon (Palma et al., 2018) to a depth of 230 mm. The model required initial inputs such as the tree planting density and the initial biomass of the tree and crops. The process of using the model, initially requires the calibration of the model outputs against measured yields from arable and tree-only systems. The Yield-SAFE model was then used to predict the tree and crop growth in an agroforestry system using seven state equations expressing the temporal dynamics of: (1) tree biomass; (2) tree leaf area; (3) number of shoots per tree; (4) crop biomass; (5) crop leaf area index; (6) heat sum; and (7) soil water content. The productivity of each system was assessed over an assumed tree rotation for fast-growing poplar (*Populus spp.*) of 30 years at the UK site, and 60 years for cherry (*Prunus avium*) and holm oak (*Quercus ilex L.*) at the Swiss and Spanish sites.

2.3 Financial analysis

The financial performance of the different land-use systems was compared in terms of their annual net margins using the Farm-SAFE bio-economic model (Graves et al., 2011). The financial data were collated from management handbooks, farmers and advisors, and previous studies (Graves et al., 2007). The resulting information for each land use was stored within a Farm-SAFE worksheet to provide a default financial dataset for each site. The collated data included crop input costs, along with tree data such as establishment, weeding and pruning (Appendix C; Table C1 and Table C3) and the levels of governmental support (Appendix C; Table C2) were assumed for a tree rotation of 30 (UK) or 60 years (Spain and Switzerland). The net margins for the arable, agroforestry, and tree-only systems were determined as the revenues from harvested products including any available grants minus the variable and assignable fixed costs. The financial net margins were then expressed as a net present value (NPV_F) (Equation 1) to account for the opportunity cost of capital and the preference that people have for money in the present rather than in the future. Thus:

$$\mathrm{NPV_F} = \sum_{t=0}^{n} \left(\frac{(R_t - VC_t - AFC_t)}{(1+i)^t} \right) \qquad Eq.1$$

where R_t, VC_t, and AFC_t are respectively revenue, variable costs, and assignable fixed costs in year t (€ ha^{-1}), i is the discount rate, and n is the time horizon for the analysis. The EU recommended reference discount rate of 4% for long term projects was chosen. The income from each system was calculated in terms of a financial equivalent annual value (EAV_F: € ha^{-1} y^{-1}) using Equation 2:

$$\mathrm{EAV_F} = \mathrm{NPV_F} * \left(\frac{(1+i)^n}{(1+i)^n - 1} \right) * i \qquad Eq.2$$

2.4 Modelling the environmental externalities

The environmental externalities were modelled using the approach described by García de Jalón et al., (2018b) and the main assumptions are repeated here for clarity. The five externalities studied were the regulation of carbon dioxide emissions, carbon sequestration, soil erosion by water, nitrogen losses, and phosphorus losses.

2.4.1 Carbon dioxide (CO_2) emissions

Annual CO_2 emissions ($Emi.CO_2$; units: t CO_2 ha^{-1} y^{-1}) for each land use system were determined by integrating a life cycle assessment (Williams et al., 2010) into the Farm-SAFE model. The emissions included were those that relate to the manufacture of fertilizer (M_F), pesticides (M_P) and field machinery (M_M), and the fuel used for cultivation (F_C), fertilizer and agrochemical application (F_F), sowing (F_S), and harvesting (F_H) (Equation 3). This analysis did not consider CH_4 and N_2O.

$$Emi.CO_2 = M_F + M_P + M_M + F_C + F_F + F_S + F_H \qquad Eq.3$$

Machinery operations were assumed to be similar for the arable system and the crop component of the agroforestry system, and for the tree-only system and tree component of the agroforestry system. Emissions from manufacture of machinery were estimated from the life expectancy of the machinery (Nix 2017) based on a per hectare utilisation rate. Field diesel, fertiliser, and pesticides emissions from manufacturing were calculated on a per hectare basis. Similarly, emissions to the atmosphere from field diesel, fertiliser, and pesticides were determined. A 'cradle-to-field gate' approach was applied i.e. emissions associated with grain drying, crop storage, and downstream processing were excluded.

2.4.2 Carbon sequestration

The annual amount of carbon sequestered by each system (*Total seq. C*; units: t C ha^{-1} y^{-1}) was calculated from the carbon stored in the timber (*Timber*), branchwood (*Branchwood*) and roots (*Roots*) (referred to, as biomass carbon), and the soil component (*Soil*) which was determined to a depth of 230 mm from the break-down of roots and leaves from trees, crops and grass (referred to, as soil carbon; Equation 4). Timber and branchwood carbon inputs were estimated from the tree growth simulations derived using Yield-SAFE. The changes in soil carbon over time were determined using the RothC model integrated into Yield-SAFE, which splits soil organic carbon into four active compartments and a small amount of inert organic matter (IOM) which is resistant to decomposition. The four active compartments are Decomposable Plant Material (DPM), Resistant Plant Material (RPM), Microbial Biomass (BIO) and Humified Organic Matter (HUM). Each compartment was assumed to decompose by a first-order process with its own characteristic rate (Coleman and Jenkinson 2014). Leaf carbon inputs were simulated by considering that leaf fall occurred over a 30-day period of each year with a leaf fall rate ranging from 0 (evergreen) to 1 (deciduous). Carbon as root biomass was assumed to equal 25% of the timber biomass (IPCC, 1996). Soil carbon inputs from the arable crop were based on the dry matter of straw left after harvest.

$$Total\ Seq.C = (Timber + Branchwood + Roots) + Soil \qquad Eq.4$$

2.4.3 Soil erosion losses by water

In order to calculate the annual soil loss by water (A_t; units t ha^{-1} y^{-1}) the Revised Universal Soil Loss Equation (RUSLE) was used within the Farm-SAFE model (Equation 5):

$$A = RKLSCP \qquad Eq.5$$

where *R* is the rainfall-runoff erosivity factor, *K* is soil erodibility, *L* is slope length, *S* is slope steepness, *C* relates to cover-management, and *P* relates to support practice that reduces the erosion potential of runoff (Table 2). The *R*, *K*, *L* and *S* values, determined by climatic, soil and topographic characteristics, were obtained from the European Soil Data Centre (ESDAC) database and the Swiss environmental department for the geographical location of the case study areas (see Panagos et al., 2014, 2015a, b and Prasuhn et al., 2007).

Table 2. RUSLE factors in terms of rainfall-runoff erosivity (*R*), soil erodibility (*K*), slope length (*LS*), cover management (*C*) and support practices (*P*) as acquired from ESDAC, Prasuhn et al., 2007 and the Yield-SAFE model

Case study	*R*	*K*	*LS*	*C*	*P*

Bedfordshire, UK	253.0	0.03	1.40	W:0.20	B:0.21	O:0.28	1
Extremadura, Spain	518.5	0.03	1.50	T:0.40	G:0.20*		1
Schwarzbubenland, Switzerland	900.0	0.03	1.45	O:0.28	W:0.20	G:0.20*	1

W: wheat, B: barley, O: oilseed rape, T: oats, G: grass

*In the agroforestry system a *C* factor of 0.17 was used for the perennial grass (Wischmeier and Smith, 1978)

The dynamic change in the cover-management factor (C_t) in year *t* was calculated for each system using equation 6 (Palma et al., 2007):

$$C_t = Cov_{crop,t}C_{crop} + Cov_{tree,t}C_{tree} \qquad Eq.6$$

where, $Cov_{crop,t}$ is the proportion of cropped land in year *t*, C_{crop} is the cover-management factor of the crop component, $Cov_{tree,t}$ is the proportion of land under the tree component in year *t*, and C_{tree} is the cover-management factor of the tree component. The *P* factor for the three land uses was obtained from the ESDAC database (Panagos et al., 2015c). Our approach considered the distance between tree lines as in Palma et al., (2007) and the changes in land cover fraction over time as a result of tree canopy growth.

2.4.4 Nitrogen balance

As described in García de Jalón et al., (2018b) the annual nitrogen balance (N_{bal}; units: kg N ha^{-1} y^{-1}) of each land use system was determined using Palma et al., (2007) and Feldwisch et al., (1998) (Equation 7):

$$N_{bal} = N_{fert} + N_{Adep} + N_{fix} + N_{min} - D - V - U - I \qquad Eq.7$$

where N_{fert} is the addition of nitrogen fertiliser, N_{Adep} is the atmospheric deposition of nitrogen, N_{fix} is the biotic nitrogen fixation, N_{min} is the mineralization of nitrogen in the soil, *D* is the denitrification, *V* is the volatilisation, *U* is the crop and tree retention and *I* is the immobilisation (all units in kg N ha^{-1} y^{-1}). The details on nitrogen balance calculations along with the assumptions regarding nitrogen fertilisation (N_{fert}) are presented in Appendix B.

2.4.5 Phosphorus balance

Annual phosphorus balance (P_{bal}; units: kg P ha^{-1} y^{-1}) was calculated from Equation 8 which shows the P inputs and outputs considered in the analysis.

$$P_{bal} = P_{fert} + PA_{dep} - U \qquad Eq.8$$

P_{fert} refers to the addition of phosphorus fertiliser, PA_{dep} to the atmospheric deposition and *U* is the crop and tree P retention (kg P ha^{-1} y^{-1}). Phosphorus fertilisation (P_{fert}) is presented in Appendix B. A 0.33 kg P ha^{-1} y^{-1} atmospheric deposition (PA_{dep}) was assumed (Tipping et al., 2014). A content of 0.2% and 0.08% P in the grain and residue was assumed, respectively (Sandaña and Pinochet, 2014). A 0.04% concentration of P in the tree biomass was also considered (Ovington and Madgwick, 1958).

2.4.6 Economic analysis

Whilst financial analysis determines the profitability from a farmer's perspective, economic analysis can determine the benefit from a societal perspective. The economic appraisal built upon the NPV_F (see Equation 1) and included benefits and costs from the five environmental externalities converted into monetary terms (EE_t) in each year *t*. The NPV for the economic appraisal (NPV_E; equation 9) was determined as:

$$\mathrm{NPV_E} = \sum_{t=0}^{n}\left(\left(\frac{(R_t - VC_t - FC_t)}{(1+i)^t}\right) + \left(\frac{EE_t}{(1+j)^t}\right)\right) \qquad Eq. 9$$

where *j* is the assumed discount rate for environmental costs and benefits (which was assumed to be 4% as in the financial analysis). From the NPV_E, the economic EAV (EAV_E) was calculated as in Equation 2.

2.4.7 Valuation of the environmental externalities - Sensitivity analysis

The sensitivity of each land use system to the value of the environmental externalities was determined by identifying the environmental externality value (€ $unit^{-1}$) at which the EAV_E of the agroforestry and tree-only systems matched the EAV_E of the corresponding arable system. In order to find the carbon value, for example, the other non-carbon externalities were set to zero. Thus, by increasing the carbon value, land-use systems that emit carbon (negative carbon sequestration) have an increasingly negative EAV_E relative to systems that sequester carbon. The value of EAV_E for each land use was the sum of each environmental externality and the systems' financial performance. The sensitivity analysis using current values for environmental externalities was also used to compare the three land-use systems with each other and against the financial baseline. The valuation of the environmental externalities was based on the Graves et al., (2015) non-traded values of €57.1 (t CO_2)$^{-1}$, €0.20 (kg N)$^{-1}$, €1.58 (kg P)$^{-1}$ and €6.4 (t soil sediment)$^{-1}$. Soil erosion valuation was based on Jacobs (2008), who estimated an annual off-site cost of dredging water courses in England and Wales of €12.9 million with an agricultural apportionment of 95%, giving a total cost (adjusted to 2009 prices) of €12.2 million. Thus, as Anthony et al., (2009) reported a sediment load of 1.9 million t yr^{-1}, a unit cost of removal of around €6.41 t^{-1} sediment was estimated.

3. Results

3.1 Biophysical simulation of crop yields and timber biomass

In the UK case study, the predicted mean yields over 14 years in the agroforestry system for wheat, barley and oilseed were 7.7, 6.0 and 3.1 t ha^{-1} respectively (Table 3). These represented mean yield reductions of 17, 10 and 8% respectively compared to the mean yield of the arable system. The Spanish arable rotation yielded 2.1 t ha^{-1} for the oats and 1.3 t ha^{-1} for the grass, which was 0.3 t ha^{-1} greater than the predicted yields of the agroforestry system. The predicted grass yield in the Swiss agroforestry system (4.4 t ha^{-1}) was 36% of the grass-yield (12.4 t ha^{-1}) in the system with no trees. The volume of the standing timber for the UK poplar plantation reached 219 m^3 ha^{-1} in year 30, while the Spanish and Swiss tree-only systems resulted in 51 and 117 m^3 ha^{-1} in year 60 respectively.

Table 3. Average annual crop yields (t ha^{-1}) of the arable (A) and agroforestry (AF) systems and standing timber volume (m^3 ha^{-1}) of the Tree-only (T) system, in the three case study sites

	UK			Spain			Switzerland		
	A	AF	T	A	AF	T	A	AF	T
Wheat	8.7	7.7[a]		-	-		5.6	-	
Barley	6.7	6.0[a]		-	-		-	-	
Oilseed	3.5	3.1[a]		-	-		3.0	-	
Oats	-	-		2.1	-		-	-	
Grass	-	-		1.3	1.0		12.4	4.4	
Standing timber *	-	216	219	-	15.6	51	-	130	117

*UK: *Populus spp.* in year 30, Spain: *Quercus ilex L.* in year 60, Switzerland: *Prunus avium* in year 60

[a] AF crop yields in the UK were for 14 years only

3.2 Financial analysis

In the UK system, without public grants, the financial net margin of the arable system expressed as a net present value (NPV_F) over 30 years (4% discount rate) was €5,444 ha^{-1} (Table 4), compared to €3,669 ha^{-1} and €1,197 ha^{-1} for the agroforestry and tree-only systems respectively. The corresponding equivalent annual values (EAV) followed a similar pattern, with the arable system resulting in the greatest value (€315 ha^{-1} y^{-1}) followed by the agroforestry (€212 ha^{-1} y^{-1}) and the tree-only system (€69 ha^{-1} y^{-1}). By including grants, the NPV_F for the arable system increased to €9,674 ha^{-1} and the agroforestry rose to €7,899 ha^{-1}. No change was observed with the tree-only system (€1,197 ha^{-1}) as for the UK as it was assumed that tree grants were not available at a tree density below 400 trees ha^{-1} (Table 4).

In the Spanish system and without accounting for grants, the arable system resulted in the highest NPV_F over 60 years (€4,635 ha^{-1}) whilst the tree-only system (Table 4) resulted in a loss (-€933 ha^{-1}). The agroforestry dehesa system showed an intermediate value of €1,952 ha^{-1}. When including grants, the NPV of each system was positive. The NPV_F of the arable system increased to €8,109 ha^{-1}, the agroforestry system reached €4,500 ha^{-1}, whilst the NPV_F of the tree-only system increased by €2,014 ha^{-1} to a value of €1,081 ha^{-1}. The effect of adding grants was to increase the EAV_F of the arable, agroforestry, and tree-only systems by €153 ha^{-1} y^{-1}, €113 ha^{-1} y^{-1}, and €89 ha^{-1} y^{-1} respectively (Table 4).

In the Swiss system without grants, the agroforestry system which had high labour and machinery costs resulted in a negative NPV_F and EAV_F. Although the NPV_F of the Swiss systems was calculated over 60 years, compared to 30 years for the UK systems, with the inclusion of grants (Appendix C; Table C2) the Swiss arable and agroforestry systems were the most profitable (Table 4). The arable resulted in a cumulative net margin in year 60 of €50,279 ha^{-1}, followed by the marginally lower agroforestry at €44,377 ha^{-1} while the tree-only system (which did not receive any governmental support) gave a negative cumulative net margin of -€1,086 ha^{-1} (Table 4). The values of the EAV_F for the Swiss arable and agroforestry systems with grants were at least four times greater than that observed in the UK.

Table 4. Financial present value of the net margin over 30 or 60 years and the equivalent annual values (EAV_F) at a discount rate of 4% for three land uses in each of three case studies: without and with government grants

Case study	Duration (y)	Land use	Without grants		With grants	
			Net margin (€ ha^{-1})	EAV_F (€ ha^{-1} y^{-1})	Net margin (€ ha^{-1})	EAV_F (€ ha^{-1} y^{-1})
Bedfordshire	30	Arable	5,444	315	9,674	559
		Agroforestry	3,669	212	7,899	457
		Tree-only	1,197	69	1,197	69
Extremadura	60	Arable	4,635	205	8,109	358
		Agroforestry	1,952	86	4,500	199
		Tree-only	-933	-41	1,081	48
Schwarz-Bubenland	60	Arable	19,481	861	50,279	2,222
		Agroforestry	-31,784	-1,404	44,377	1,961
		Tree-only	-1,086	-48	-1,086	-48

In terms of the cash flow profile, the cumulative net margin of the UK agroforestry system without grants remained similar between years 15 and 29, as arable cropping stopped once the tree canopy closed (Appendix A: Interactive Figure A1). The Spanish tree-only system was unprofitable without grants over the 60 years but including grants was more profitable than arable and agroforestry in year 2 and 5 respectively (Appendix A: Interactive Figure A1). In Switzerland the tree-only and agroforestry systems were unprofitable without grants whereas the arable system was still profitable. With grants, the Swiss agroforestry system showed its lowest NPV_F for the period up to year 8, but the system gradually became more profitable due to the revenue from cherry production (assuming constant cherry yields after year 10, Appendix A: Interactive Figure A1).

3.3 Environmental externalities

In each case study, the assumed carbon dioxide emissions from the tree-only systems were negligible apart from the use of machinery for tree planting in the first year and tree cutting in the final year (Figure 1). The calculated annual emissions in the arable systems varied around a consistent mean value during the 30- or 60-year period, ranging from 1.1 t CO_2 ha^{-1} at the Swiss site to 2.4 t CO_2 ha^{-1} at the UK site (Figure 1; Table 6). The British agroforestry system, where arable cropping stopped after year 14, resulted in a lower mean annual carbon dioxide emission of 1.3 t CO_2 ha^{-1} over the 30 years, compared to the arable system. Similarly, in Switzerland as the trees matured, emissions from the agroforestry system became lower than the arable.

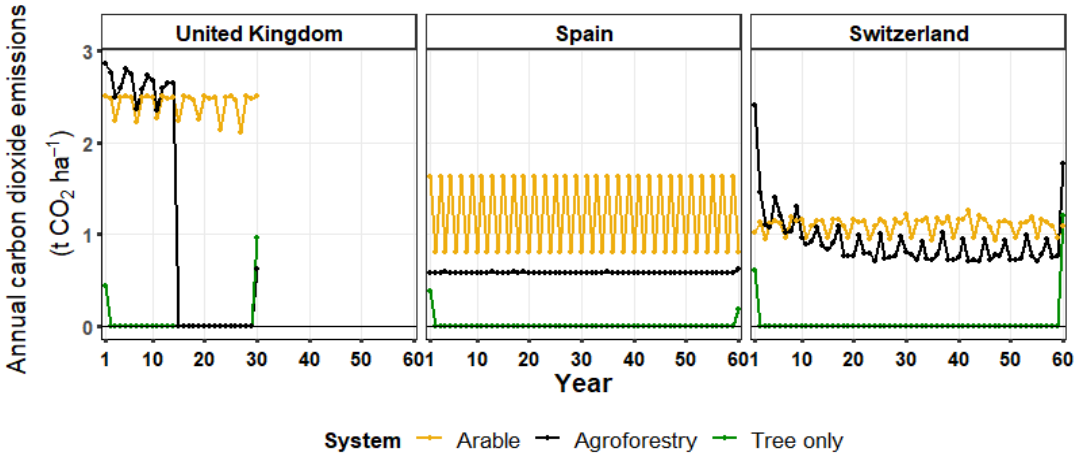

Figure 1. Modelled annual CO_2 emissions for the arable, agroforestry, and tree-only for UK, Spain, and Switzerland over 30, 60, and 60 years respectively

The dehesa agroforestry system in Spain resulted in lower average emissions (0.6 t CO_2 ha^{-1} y^{-1}) than the arable (1.2 t CO_2 ha^{-1} y^{-1}) system during the 60 years of the analysis. The relatively low CO_2 emissions in the dehesa system is due to it being a silvopastoral system with low machinery use.

Across the three case studies, the potential change in carbon storage within the arable systems was limited to soil carbon whereas the changes in the agroforestry and tree only systems included both tree biomass and soil carbon. The level of soil carbon to a depth of 230 mm in the UK (20.7 t C ha^{-1}) and the Swiss (20.3 t C ha^{-1}) arable systems remained relatively constant, whereas the soil carbon declined in the Spanish arable system from 20.7 t C ha^{-1} in year 1 to 13.7 t C ha^{-1} in year 60 (Figure 2). Higher levels of total carbon storage were modelled in the agroforestry and tree-only systems. The lowest level of total C storage in the tree-only systems occurred in Spain being 18 t C ha^{-1} in year 30 and 49 t C ha^{-1} in year 60 (Figure 2). The carbon storage in the UK tree-only system was 104 t C ha^{-1} in 30 years.

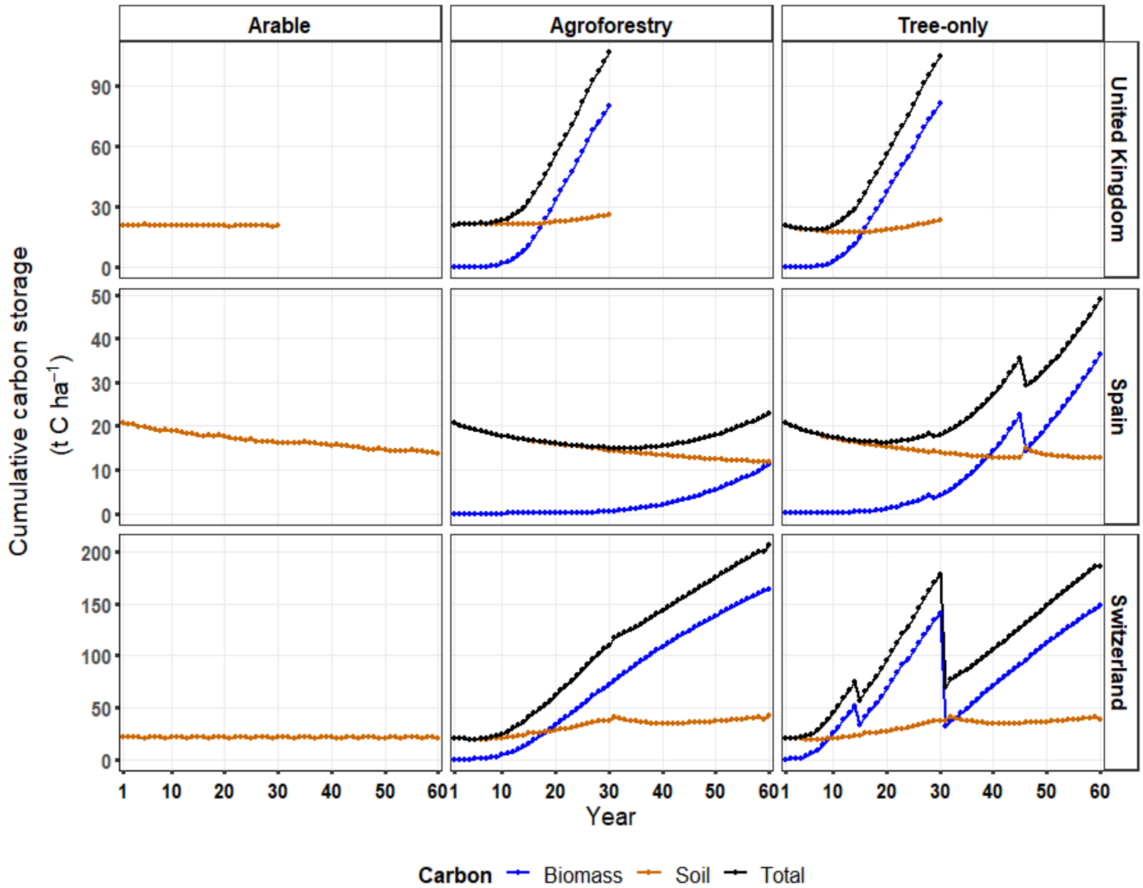

Figure 2. Modelled carbon storage (t C ha^{-1}) as: biomass (above and belowground), soil (which also includes fallen leaf carbon) and in terms of total carbon (biomass plus soil). In the arable system, the soil carbon is the same as the total carbon as the biomass carbon was assumed to be zero. Note that the y-axes have different ranges.

The Swiss tree-only system stored 178 t C ha^{-1} by year 30 and 186 t C ha^{-1} by year 60 (Figure 2). These values exclude any carbon stored in the thinnings, which were assumed to be rapidly lost back to the atmosphere due to decay if left in the field or through combustion if used as firewood. When the trees in the Swiss system were thinned, the soil carbon was assumed to decrease due to less leaf matter and small branches falling on the ground. Total carbon accumulation for the agroforestry systems in year 30 were 106 t C ha^{-1} for the UK system, 15 t C ha^{-1} for the dehesa system (23 t C ha^{-1} over 60 years), while the Swiss agroforestry system sequestered 109 t C ha^{-1} (206 t C ha^{-1} over 60 years; Figure 2).

The cumulative net carbon sequestration of each system (Figure 3) was calculated by combining the cumulative net CO_2 emissions (Appendix D; Figure D1) with the cumulative sequestered carbon (Figure 2). The negative net carbon sequestration of the arable system in each country resulted in the net emission of carbon to the atmosphere. In the tree-only systems, the Swiss system resulted in a sink of 332 t CO_2 ha^{-1} between year 1 and year 60, the UK system provided a sink of 157 t CO_2 ha^{-1} over 30 years, and the Spanish system had a cumulative net carbon benefit of 37 t CO_2 ha^{-1} over 60 years. The agroforestry values were between the arable and tree-only systems, with the UK system providing a sink of 127 t CO_2 ha^{-1} over 30 years, the Spanish system resulted in a net emission of 47 t CO_2 ha^{-1} over 60 years, while the Swiss agroforestry system resulted in similar sink to the Swiss tree-only system of 326 t CO_2 ha^{-1} over 60 years.

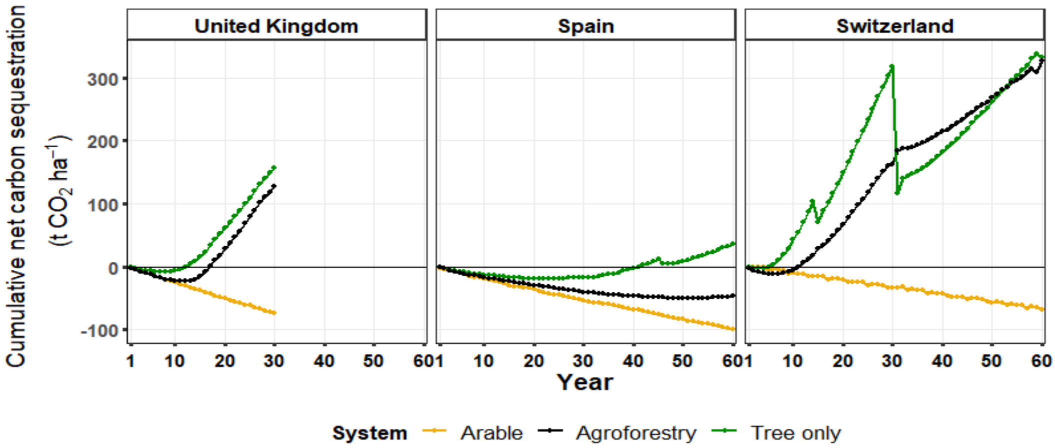

Figure 3. Modelled cumulative net carbon sequestration for the arable, agroforestry and tree-only systems for UK, Spain, and Switzerland over 30, 60, and 60 years respectively.

The calculated mean annual rate of soil loss due to water from the arable system in the UK was 2.1 t ha^{-1} (Table 6; Appendix D: Figure D2). The mean annual rate of soil loss in Spain ranged from 4.2 t ha^{-1} for grass to 8.4 t ha^{-1} for the oats; the mean annual rate over the 60 years was 6.3 t soil ha^{-1}. In Switzerland, the high levels of rainfall and long slope lengths resulted in high annual rates of soil erosion ranging from 7.8 t ha^{-1} for the wheat-grass-wheat component to 10.9 t ha^{-1} for oilseed rape (Appendix D: Figure D2). The mean annual rate over 60 years was 8.6 t soil ha^{-1}. In each country the addition of trees reduced soil erosion (Appendix D: Figure D2). For example, the mean annual rate of soil loss by water for the UK was 0.9 t ha^{-1} for the poplar plantation and 1.0 t ha^{-1} for the agroforestry over 30 years, with the rate declining (as the trees mature) to below 1.0 t ha^{-1} in year 14 and 15 respectively.

The cumulative nitrogen (N) and phosphorus (P) balances were greater in the arable systems than in the tree-only and agroforestry systems in each country. Over 30 years, the arable nitrogen balance ranged from 367 to 1437 kg N ha^{-1} and the phosphorus from 103 to 365 P ha^{-1} (Table 5). By contrast the tree-only systems resulted in a net uptake of between 637 and 1,718 kg N ha^{-1} and 663 to 1,257 kg P ha^{-1}. The agroforestry systems allowed the continued production of food with an intermediate nutrient balance (Table 5).

Table 5. Cumulative nitrogen (N) and phosphorus (P) balances (kg ha^{-1}) from year 1 to year 30 for the arable, agroforestry, and tree-only systems at the case study sites in the UK, Spain and Switzerland

Parameter	Country	Arable	Agroforestry	Tree-only
N balance	UK	1,250	-193	-1,469
	Spain	367	-453	-637
	Switzerland	1,437	812	-1,718
P balance	UK	365	-50	-1,181
	Spain	103	4	-663
	Switzerland	341	35	-1,257

3.4 Valuation of the environmental externalities

The next stage was to compare the societal benefit of the land-use systems by including the economic value of the environmental externalities (Table 6). For the arable systems, the greatest societal cost was associated with carbon dioxide emissions equivalent to €144 ha^{-1} y^{-1} in the UK, €73 ha^{-1} y^{-1} in Spain, and €64 ha^{-1} y^{-1} in Switzerland. The arable systems also resulted in substantial soil erosion costs in Spain (€43 ha^{-1} y^{-1}) and Switzerland (€58 ha^{-1} y^{-1}). In the tree-only systems, the greatest positive benefit in the UK and Switzerland was associated with carbon sequestration with values of €227 ha^{-1} y^{-1} and €341 ha^{-1} y^{-1} respectively (Table 6). The annual carbon sequestration benefit of the Spanish tree-only system was marginally below zero (-€11 ha^{-1} y^{-1}) because of the soil carbon losses.

At the British site, the benefits of carbon sequestration combined with the low rate of CO_2 emissions resulted in the tree-only system providing a greater societal benefit (€285 ha^{-1} y^{-1}) than the agroforestry (€137 ha^{-1} y^{-1}) and the arable system (-€190 ha^{-1} y^{-1}) (Table 6). In Spain, the societal benefit of the tree-only system (in terms of the five externalities examined) was only €2 ha^{-1} y^{-1}, but this was still greater than that of the dehesa agroforestry (-€93 ha^{-1} y^{-1}) and the arable system (-€154 ha^{-1} y^{-1}). Over 60 years in Switzerland, the soil erosion costs associated with the agroforestry system (€43 ha^{-1} y^{-1}) was marginally less than that with the arable (€58 ha^{-1} y^{-1}) and the tree-only land use (€48 ha^{-1} y^{-1}). However, the overall societal benefit of the agroforestry system (€206 ha^{-1} y^{-1}) was between that of the arable (-€152 ha^{-1} y^{-1}) and the tree-only system (€354 ha^{-1} y^{-1}).

Table 6. Financial, economic and environmental externalities equivalent annual value (EAV; discounted all at 4%) of an arable (A), agroforestry (AF), and tree-only (T) system in the UK, Spain and Switzerland

	United Kingdom			Spain			Switzerland		
Financial analysis	A	AF	T	A	AF	T	A	AF	T
EAV_F with grants (€ ha^{-1} y^{-1})	559	457	69	358	199	48	2,222	1,962	-48
EAV_F without grants (€ ha^{-1} y^{-1})	315	212	69	205	86	-41	861	-1,405	-48
Environmental externalities									
CO_2 emissions (t CO_2 ha^{-1} y^{-1})	2.4	1.3	0.0	1.2	0.6	0.0	1.1	0.9	0.0
EAV CO_2 emissions (€ ha^{-1} y^{-1})	-144	-96	-2	-73	-34	-1	-64	-60	-2
CO_2 sequestration (t CO_2 ha^{-1} y^{-1})	0.0	5.5	5.3	-0.4	-0.2	0.6	0.0	6.3	5.6
EAV CO_2 sequestration (€ ha^{-1} y^{-1})	-3	247	227	-30	-35	-11	-1	318	341
Soil erosion (t soil loss ha^{-1} y^{-1})	2.1	1.0	0.9	6.3	4.0	3.6	8.6	6.1	5.0
EAV soil erosion losses (€ ha^{-1} y^{-1})	-14	-9	-7	-43	-27	-33	-58	-43	-48
Nitrogen balance (kg N ha^{-1} y^{-1})	41.6	-6.4	-49.0	12.2	-15.2	-23.8	49.6	8.6	-52.6
EAV nitrogen balance (€ ha^{-1} y^{-1})	-9	-1	9	-2	3	4	-10	-5	10
Phosphorus balance (kg P ha^{-1} y^{-1})	12.1	-2.1	-39.3	3.5	0.1	-23.4	11.6	2	-32.6
EAV phosphorus balance (€ ha^{-1} y^{-1})	-20	-2	58	-6	0	37	-19	-4	53
Sum EAV of environmental externalities	-190	137	285	-154	-93	2	-152	206	354
Economic analysis									
EAV_E with grants (€ ha^{-1} y^{-1})	369	596	353	204	106	44	2,072	2,167	306
EAV_E without grants (€ ha^{-1} y^{-1})	125	351	353	51	-7	-45	709	-1,199	306

3.5 Sensitivity analysis

One of the advantages of developing an economic model is the ability for the user to determine the sensitivity of the outputs to specific inputs. Thus, the next step was to identify the societal price for each of the four environmental externalities (assuming a zero price per unit for the other externalities) at which the societal equivalent annual value (EAV_E) fulfilled two scenarios (Table 7). These were: 1) the societal EAV of agroforestry to match that of arable ($EAV_{E\_Agroforestry} = EAV_{E\_Arable}$) and 2) the societal EAV of the tree-only to match that of the arable ($EAV_{E\_Tree\_only} = EAV_{E\_Arable}$). Any prices greater than those in Table 7, per environmental externality, would result in agroforestry or tree-only systems being more profitable than the arable. The analysis was completed for both "with" and "without" governmental support in terms of grants per country. Because grants represent a transfer of money from one part of society to another, the societal benefit of the systems is most clearly demonstrated without grants.

In the UK and Spain, without grants, the agroforestry systems provided a more or equally cost-effective way as the tree-only systems for controlling soil erosion (per tonne of soil) and reducing nitrogen losses (per kg of N) on arable land (Table 7). By contrast, in Switzerland, the tree-only system (which didn't receive any governmental support) was a more cost-effective way of reducing each of the four externalities than agroforestry. In all countries and scenarios, the tree-only system was a more cost-effective way of controlling phosphorus losses than agroforestry when not accounting for grants (Table 7).

In terms of carbon prices, British agroforestry required a lower price (per tonne of carbon dioxide) both without grants and with grants (€16 per t CO_2) than the tree-only system to match the arable EAV (Table 7; Appendix C: Table C4). By contrast, without grants, the Spanish and Swiss agroforestry systems required a higher value for carbon than that required for the tree-only systems to equalize the EAV of arable (Table 7; Appendix C: Table C4).

Table 7. Identified societal values per environmental externality and country in order for the economic equivalent annual value (EAV) of agroforestry (Scenario 1: $EAV_{E\_Agroforestry} = EAV_{E\_Arable}$) and tree-only systems to match that of the arable system (Scenario 2: $EAV_{E\_Tree\text{-}only} = EAV_{E\_Arable}$)

Case study	Scenario	Grants	Carbon	Nitrogen	Phosphorus	Erosion
			€ (t CO_2)$^{-1}$	€ (kg N)$^{-1}$	€ (kg P)$^{-1}$	€ (t soil)$^{-1}$
United Kingdom	1	With	16	3	8	95
	2	With	64	6	10	403
	1	Without	16	3	8	95
	2	Without	32	3	5	202
Spain	1	With	185	6	47	67
	2	With	137	9	12	113
	1	Without	137	4	35	50
	2	Without	109	7	9	89
Switzerland	1	With	40	7	19	102
	2	With	340	22	51	626
	1	Without	345	55	161	885
	2	Without	136	9	21	251

$EAV_{E\_Agroforestry} = EAV_{E\_Arable}$: Environmental externality price at which the equivalent annual value of agroforestry matches that of the arable system

$EAV_{E\_Tree\text{-}only} = EAV_{E\_Arable}$: Environmental externality price at which the equivalent annual value of the tree-only system matches that of the arable system.

Figure 4 illustrates the individual effects of changes in the unit value of the carbon, nitrogen, phosphorus and soil erosion externalities on EAV_E (assuming no grants) of the arable, agroforestry, and tree-only systems in the UK. The black solid line shows their combined effect. Corresponding graphs for the Spanish and Swiss case studies are presented in Appendix A (Interactive Figure A2). For the UK, the societal EAV is most sensitive to a relative change in the price of carbon assuming a default price of €57 (t CO_2)$^{-1}$. The effect of changes in the assumed values of nitrogen and phosphorus pollution (assuming default prices of €0.20 (kg N)$^{-1}$ and €1.58 (kg P)$^{-1}$) were relatively small. In Spain and Switzerland, the effect of changes in the carbon price was also important along with erosion for Spain (Appendix A: Interactive Figure A2).

Increasing the societal price of carbon reduced the arable EAV_E in each of the three case study countries, with the greatest reduction noted in the UK system. In the UK arable system, a 100% increase in the carbon value from a default value of €57 (t CO_2) $^{-1}$, reduced the EAV_E of the system to €34 ha^{-1} y^{-1} (Figure 4). By contrast, the high carbon sequestration rate of the tree-only system, meant that the EAV_E increased from €62 ha^{-1} y^{-1} (no carbon value included) to €667 ha^{-1} y^{-1} at a carbon value of €114 (t CO_2)$^{-1}$ (Figure 4; Tree-only-green line). The same increase of the societal value of carbon increased the EAV_E of agroforestry from €212 ha^{-1} y^{-1} to €697 ha^{-1} y^{-1} (Figure 4). In terms of soil erosion, the profitability of each system reduced when the relative cost per tonne of sediment changed from zero (e.g. -100%) to €12.8 t^{-1} (e.g. +100%). The UK arable system showed the greatest EAV_E reduction (€27 ha^{-1} y^{-1}), followed by the agroforestry (€13 ha^{-1} y^{-1}) and tree-only system (€11 ha^{-1} y^{-1}). In the UK, the effect of the societal value of erosion on EAV was less, compared to Spain and Switzerland because the UK area was relatively flat and received the lowest amount of rainfall (Table 2; Appendix A: Interactive Figure A2).

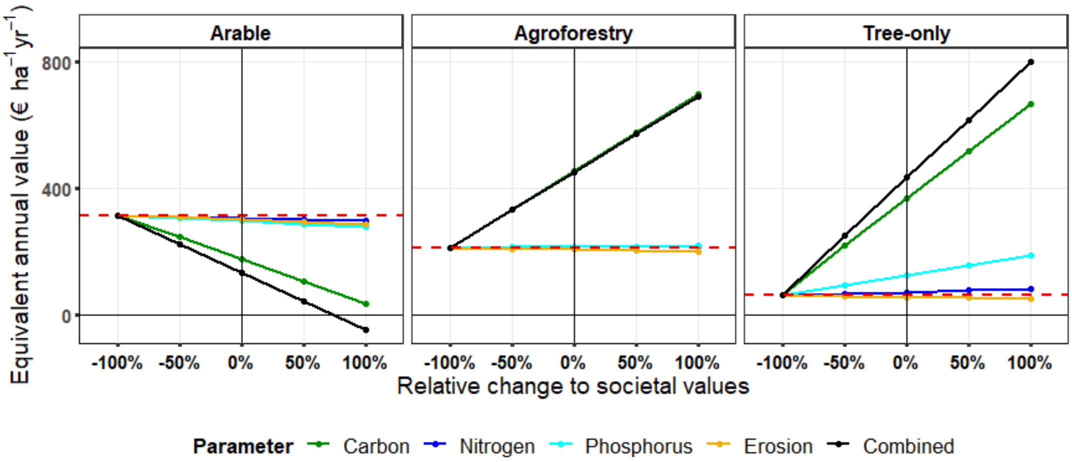

Figure 4. Sensitivity analysis on the equivalent annual value for the United Kingdom per land use system (when not accounting for grants) as affected by the value of environmental externalities relative to the default prices of €57.1 (t CO_2)$^{-1}$, €0.20 (kg N)$^{-1}$, €1.58 (kg P)$^{-1}$ and €6.4 t^{-1} soil sediment. The red dotted lines correspond to the financial performance (baseline) of the associated system.

Within the Spanish system the financial profitability of the arable baseline without grants was €205 ha^{-1} y^{-1} and by introducing the societal value of carbon, the EAV reduced to €16 ha^{-1} y^{-1} (Appendix A: Interactive Figure A2). The Spanish system, was also sensitive to the assumed cost of soil erosion, and hence the EAV_E of the arable, agroforestry and tree-only reduced by €81 ha^{-1} y^{-1}, €50 ha^{-1} y^{-1} and €46 ha^{-1} y^{-1} respectively when the erosion societal value increased from €6.4 t^{-1} to €12.8 t^{-1}. Finally, in Switzerland the arable system when not accounting for grants, also showed a decrease in EAV_E when carbon value was integrated into the economic modelling, from €861 ha^{-1} y^{-1} to €731 ha^{-1} y^{-1}. In tree-only and agroforestry however, the EAV increased from €-48 ha^{-1} y^{-1} to €585 ha^{-1} y^{-1} and from €-1,405 ha^{-1} y^{-1} to €-783 ha^{-1} y^{-1} respectively (Appendix A: Interactive Figure A2).

4. Discussion

The above results are discussed in terms of the value of carbon, the cost of excess nitrogen and phosphorus, soil erosion, the implications for agri-environmental measures, and the limitation of the study.

Value of carbon

Increasing tree cover on agricultural land can substantially enhance carbon sequestration particularly in terms of biomass carbon storage. However, at a global level, the need to sequester carbon must also occur alongside the continued production of food (Foley et al., 2011). Agroforestry provides one means of increasing carbon sequestration whilst maintaining food production. This may be through the use of fruit trees, as in the Swiss system, the continued cropping of understorey arable crops as in the British case study or the grazing of cows as in the Spanish system. The complementary use of light, nutrient and water resources by the tree and understorey components of an agroforestry system can enable absolute rates of production per hectare to be greater than separate agricultural and tree-only systems (Graves et al., 2009; Li et al., 2013).

A comparison of the benefits and costs of different land use systems can be undertaken from a financial perspective, e.g. a focus on farm profitability, or from an economic perspective that includes societal value for the principal environmental externalities. In some cases, there are practical attempts to create a market for environmental externalities. For example, farmers can benefit from markets in carbon storage, such as the Woodland Carbon Registry established by the Forestry Commission in the UK. Such initiatives allow farmers and land managers to receive credit for storing carbon, which can then be sold to other businesses wishing to offset their carbon emissions.

Excluding governmental grants, the societal benefit of the poplar agroforestry system was equal to the arable system examined in the UK (assuming the societal values for soil erosion, nitrogen and phosphorus were zero) when the value of the sequestered carbon was assumed to be at least €16 per tonne of CO_2 (Table 7; Appendix C: Table C4). This value is similar to the carbon credit value of US$ 21.7 t^{-1} of carbon (€18.9 t^{-1}) used for rice production in India (Bhola and Malhotra 2014; Nayak et al., 2019). However, these values are higher than the market price of €7.20 per tonne CO_2 for woodland planting in the UK in January 2014 (£6 at €1.20 per £1.00 exchange rate) (UK Forestry Commission 2015). Nevertheless, the UK Department of Business, Energy and Business Strategy (2018) predicts that the value of carbon will increase with a central estimate of €91 (£79) per tonne of CO_2 by 2030, and the market price on the European Union market reached €26 per tonne in May 2019 (ICE 2019). This is close to the value of €32 (t CO_2)$^{-1}$ where the tree-only system in the UK resulted in the same EAV_E as the arable system (Table 7; Figure 4; Appendix C: Table C4).

In Switzerland, a value of €345 per tonne CO_2 was needed for the EAV_E of agroforestry to match that of the arable system, whilst €136 (t CO_2)$^{-1}$ was needed for the tree-only system which received no governmental grants. In Spain, the relatively low rates of carbon sequestration by the trees and the relatively high financial value of the arable system meant that the societal value of carbon had to be €137 per tonne of CO_2 for the agroforestry system to have a similar societal EAV as the arable system. The corresponding value for the tree-only system, with a tree density of 600 trees ha^{-1}, was €109 per tonne of CO_2 (Table 7). Romanyà et al., (2000), Marcos et al., (2007), and Llorente et al., (2010) found that tree-plantations (*Pinus halepensis* Mill. and *Pinus sylvestris* L.) in Castilla y León and Vallgorguina valley in Spain needed several decades to recover the carbon lost during the process of tree establishment. This matches the modelled output indicating that the dehesa system needed more than 60 years to recover the carbon lost when introducing young oak trees on arable land.

Cost of excess nitrogen

The negative effect of excess nitrogen within agricultural systems is widely recognised, but the specific economic cost per unit nitrogen is site specific (Keeler et al., 2016). Van Grinsven et al., (2013) estimated that the societal cost in Europe of excess nitrogen on surface water in terms of eutrophication and biodiversity ranged between €5 and €20 (kg N)$^{-1}$, and Brink et al., (2011) estimated human health costs of excess nitrate entering groundwater of €0.7 (kg N)$^{-1}$. Excess nitrogen can also lead to increased emissions of the greenhouse gas nitrous oxide. By contrast van Grinsven et al., (2013) estimated that the long-term crop-yield benefits of soil nitrogen could be equivalent to €1.5 to €5 (kg N) $^{-1}$. The societal value of nitrogen per kg, needed to equalize the EAV_E of the agroforestry and arable systems, ranged from €3 to €6 (kg N)$^{-1}$ in the UK and Spain to €7-55 (kg N)$^{-1}$ in Switzerland (Table 7). These results suggest that the integration of trees with agricultural systems in the UK and Spain could be a cost-effective way of reducing nitrate levels in surface and ground water. Hence our results support the financial viability of technical initiatives to improve water quality such as the integration of trees within outdoor pig production (Manevski et al., 2019), and the creation of silvoarable or alley systems for arable crops (Wolz et al., 2018).

Cost of excess phosphorus and soil erosion

Excess phosphorus in surface water can cause environmental damage through eutrophication, and much of the flow of phosphorus is related to soil erosion and extreme rainfall events (Rodríguez-Blanco et al., 2010). Hence it is useful to consider phosphorus and soil erosion together. Estimates of the cost of reducing phosphorus in surface water in Sweden include €7 (kg P)$^{-1}$ from reducing the phosphorus intake of livestock, €220 (kg P)$^{-1}$ for afforestation of agricultural land, and €300 (kg P)$^{-1}$ for reducing livestock densities (Malmaeus and Karlsson 2010). Dockhorn (2009) reports that the cost of removing phosphorus during waste water treatment can range from €2-3 (kg P)$^{-1}$ up to €10 (kg P)$^{-1}$ under specific circumstances. In this study in the absence of grants, the value of phosphorus required for the tree-only and agroforestry systems to be financially equivalent to arable cropping ranged from €5-8 (kg P)$^{-1}$ in UK to €9-35 (kg P)$^{-1}$ in Spain (Table 7).

In a UK study, Graves et al., (2015) assumed a mean cost of soil erosion from agricultural land of €57 t^{-1}. In this study, without grants, the EAV_E of the agroforestry and tree-only system in Spain matched that of the arable system when the cost of soil erosion was €50 and €89 per tonne respectively

(Table 7). Assuming that the values for soil erosion costs for the UK are transferable to the Spanish site, then the agroforestry system would offer greater societal benefit than the arable system. The EAV_E of the agroforestry and tree-only system in the UK matched those of the arable system when their societal erosion prices were €95 t^{-1} and €202 t^{-1} respectively, suggesting that at this site, the reduction of water-based soil erosion alone was insufficient to warrant a change from arable production to agroforestry. The case study site was relatively flat and hence water-based erosion was minimal, however a more complete analysis would include the effect of trees on reducing soil erosion due to wind. In Switzerland, the agroforestry and tree-only systems required the cost of soil erosion to be €885 t^{-1} and €251 t^{-1} respectively to have an EAV_E that matched the arable system.

Implications for agri-environmental measures and policy

In the EU and the UK, there is increasing interest that public subsidies are used to provide public environmental benefits that are often undervalued by the market. Each European country supports agriculture with subsidies; in Spain and the UK at the time of this study, this is within the context of the European Union's Common Agricultural Policy. Current arrangements within the UK mean that the arable and the agroforestry systems would receive €235 ha^{-1} y^{-1} from Pillar I of the CAP. Within England, tree planting grants (linked to Pillar II) are only available for tree densities above 400 trees per hectare (UK Government 2015) i.e. at a higher density than that assumed for the agroforestry and the tree-only systems in this study. As the UK government (HM Government 2018) argues that public money provided to farmers should be targeted for the provision for non-market public services, subsidies that recognized the environmental benefits of low density trees could increase the financial attraction to farmers of the tree-only and the agroforestry systems relative to continued arable cropping.

In Spain, current agricultural subsidies also have a regressive effect in terms of environmental benefits as the externality values at which the EAV_E of the agroforestry and tree-only systems matched that of the arable system was lower without subsidy than with subsidy. The environmental impact of the three systems ranged from a cost of €154 ha^{-1} y^{-1} for arable to a benefit of €2 ha^{-1} y^{-1} for forestry. Again, a public subsidy system that paid for public benefits and imposed costs for environmental damage, would mean that the financial attraction of the tree-only and agroforestry system would improve relative to arable cropping.

The subsidy system in Switzerland is a national scheme that operates outside of the CAP and it provides particularly generous payments for agricultural production (€1,320 ha^{-1} y^{-1}) and agroforestry systems (€2,815 ha^{-1} y^{-1}), but no payment for tree-only systems. A subsidy system that accounted for only the environmental costs and benefits considered in this paper, would reduce the loss associated with the tree system and decrease the net margin of the arable system.

Limitations of the present study

It should be noted that this study included only five environmental externalities. One of the specific environmental benefits of dehesa is the biodiversity value, and if this was included, the societal benefit of the dehesa system would be greater. Hence, the inclusion of additional cultural benefits such as recreation, historical and heritage importance, and biodiversity could further strengthen the attraction of tree planting and management.

As indicated in the introduction, land use decisions depend on a range of drivers. The relative financial and economic value of alternative systems is important, but it is not the only driver (García de Jalón 2018a). Assuming that a farmer's decision is driven by financial criteria, it is still possible that a farmer will not automatically change from one land use system to another directly in response to the most profitable EAV. There are also transaction and administrative costs in changing land use, and hence greater financial inducements than those indicated may be required to ensure land use change.

The Yield-SAFE biophysical model and the Farm-SAFE economic models have been used in previous studies to predict the biophysical or economic performance of arable, agroforestry and tree-only systems (Graves et al., 2007; van der Werf et al., 2007; Sereke et al., 2015; García de Jalón 2018b). When using the models for new sites and systems, substantial work is needed to collate new parameters and financial data. One of the techniques used in the model to minimise the work required is to quantify all inputs and outputs in terms of a physical quantity (e.g. wheat yield: t ha^{-1}) and a monetary value per physical unit (e.g. value of wheat: € t^{-1}). Although the physical quantities will vary in line with the planting arrangements, the weather and the soil conditions at a specific site, the monetary values will normally be similar across a region or nation.

Although not discussed in this paper, the Farm-SAFE economic model can also be used to determine the effect of one-off changes in prices in a future year or incremental changes in prices and costs from a given future year. We recognise that any prediction is subject to uncertainty and a sensitivity analysis, as demonstrated here for environmental externalities values, is one method to examine this. The analysis also considered each externality individually. In practice, many of the externalities occur as "packages" and hence biodiverse systems such as agroforestry can offer "economies of multi-functionality" even though they may not offer the "economies of scale" of intensive arable production.

5. Conclusions

The current study presents a framework for the integrated valuation of arable, agroforestry and tree-only systems in three European case study sites, which can be applied to other locations and systems. If such land-use systems are only seen from a narrow financial perspective, and there are no incentives for farmers/land managers to implement agroforestry or tree-only systems, then their adoption can be impeded. By including societal values for environmental benefits (compared to arable), agroforestry and tree-only systems will be more highly valued. The quantification of such values provides governments and others with an approach to devise regulations or incentives that can transparently support more appropriate decision making on farms. On some farms, this will lead to tree-only and agroforestry being more attractive in specific locations. What measures are required to promote tree planting and management on farms to help achieve net zero greenhouse gas emissions? What measures are needed to protect soil or improve water quality? The answers to such questions are complicated and site specific. The study and the framework described in this paper demonstrates that it is possible to carry out whole system valuations (including environmental externalities) for contrasting systems and to identify the values that society need to assign to those externalities, to encourage selected farmers/land managers to modify land use systems.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version

Appendix B. Nitrogen balance calculations

Atmospheric deposition *(NA_{dep})* was estimated from the deposition of oxidized and reduced nitrogen from EMEP (2003). Nitrogen fixation (N_{fix}) was assumed to be 1 kg N ha^{-1} y^{-1} for non-symbiotic organisms since there was no legume crop (Wild, 1993). N mineralisation (N_{min}) and immobilisation (*I*) were assumed to reach a long-term equilibrium where the amount of mineral nitrogen released by the soil would be equal to the amount annually returned to the soil in the form of organic matter (Vlek et al., 1981; Noy-Meir and Harpaz, 1977). Denitrification (*D)* was assumed to be 30 kg N ha^{-1} y^{-1} (Palma et al., 2007). Nitrogen volatilisation (*V)* was assumed to be derived from mineral *N* application, since organic fertilisation was not considered. Following van Keulen et al., (2000), it was estimated as 5% of N_{fert}. Nitrogen retention (*U;* Equation B1*)* from the tree and the crop was estimated as:

$$U = \begin{cases} \dfrac{Y_c}{\alpha} + \lambda * \beta_t & if \quad Y_c < \dfrac{Y_{max}}{2} \\ \dfrac{4 * Y_c - Y_{max}}{2 * \alpha} + \lambda * \beta_t & if \quad Y_c \geq \dfrac{Y_{max}}{2} \end{cases} \qquad Eq.B1$$

where Y_c is the harvested crop yield, Y_{max} is the maximum harvested crop yield (kg ha^{-1} y^{-1}), and β_t is the above-ground tree biomass (kg ha^{-1} y^{-1}). The unit-less coefficient α depends on the biomass of the crop residue and the harvested product (see Equation B2). The value of *λ* is a unit-less coefficient to derive tree nitrogen retention from β_t (see Equation B3).

$$\alpha = \frac{1}{NC_c + NC_r * \frac{Y_r}{Y_c}} \qquad Eq.B2$$

where NC_c and NC_r are the N content in the crop grain and residue, respectively. A content of 2% and 0.5% N in the grain and residue was assumed respectively (Ladha et al., 2015). Y_r is the residue yield (kg ha^{-1} y^{-1}). The λ coefficient is given as:

$$\lambda = C_{tab} + C_{tbg} RSR \qquad Eq.B3$$

Where C_{tab} and C_{tbg} are the N content in the above- and below-ground tree biomass, respectively. Contents of 0.66% and 0.41% concentration of N in the tree above- and below-ground biomass were assumed respectively (Gifford, 2000 a,b). *RSR* is the root to shoot ratio of the tree (unit-less). A root to shoot ratio of 0.25 for broadleaved tree species was assumed (IPCC, 1996).

Appendix C. Assumption for costs and government subsidies, and calculated profitability,

Table C1. Assumptions for associated costs in the analysis[**]

Country	Crop	Grain price (€ t^{-1})	Seed rate (kg ha^{-1})	Fertiliser rate (kg N ha^{-1})	Fertiliser rate (kg P_2O_5 ha^{-1})	Fertiliser rate (kg K_2O ha^{-1})	Variable costs[a] (€ ha^{-1})	Fixed costs[b] (€ ha^{-1})	Labour costs (€ ha^{-1})
UK	Wheat	174[*]	160	175	60	55	653	444	38
	Barley	160[*]	155	145	55	40	535	444	38
	Oilseed	361[*]	5	200	55	45	633	444	38
Spain	Oat	159	100	250	250	250	240	114	27
	Grass	159	45	0	150	0	48	100	22
Switzerland	Oilseed	800	5	200	55	45	1462	2868	66
	Wheat	590	160	175	60	90	1182	3100	66
	Grass	355	32	0	0	0	0	1884	66

[a]Includes seed, fertiliser, spray and other costs
[b]Includes costs relating to fuel and repairs, machinery, interest on working capital, installation, rent and other fixed costs
*Nix 2017; ** Values are based on interaction with farmers, experts and end-users per case study

Table C2. Assumptions for annual governmental support as grants per country and land-use

		United Kingdom (€ ha^{-1})	Spain (€ ha^{-1})	Switzerland (€ ha^{-1})
Crop	Wheat	235	-	1,232
	Barley	235	-	-
	Oilseed	235	-	1,848
	Oat	-	187	-
	Grass	-	107	880
Tree	Tree-only Poplar	0	-	-
	Agroforestry Poplar	0	-	-
	Tree-only holm oak	-	2,013	-
	Dehesa	-	30	-
	Wild cherry Timber	-	-	0
	Wild cherry Fruit (year: 1-10)			2,182
	Wild cherry Fruit (year: 11-60)	-	-	3,449

Table C3. Summary of cumulative costs associated with the tree component of the systems

		United Kingdom		Spain		Switzerland	
Tree operations	Units	Tree only	Agroforestry	Tree only	Agroforestry	Tree only	Agroforestry
Establishment cost (total)	€ ha^{-1}	753	753	1,247	124	3,102	10,212
Costs of individual plants	€ $tree^{-1}$	2	2	0.3	0.3	0.8	55
Costs of individual tree protection	€ $tree^{-1}$	2	2	0.5	0.5	0.8	12
Costs of ground preparation	€ ha^{-1}	48.2	48.2	216	37	167	167
Labour for planting trees	min $tree^{-1}$	3	3	3	3	2	100
Labour for tree protection	min $tree^{-1}$	0.4	0.4	3	3	1	12
Weeding cost (total)	€ ha^{-1}	20.4	14.7	0	0	39	0
Annual cost of herbicide	€ $tree^{-1}$	0.02	0.02	0	0	0.22	22.5
Pruning cost (total)	€ ha^{-1}	805	805	4,970	264	2,436	2,167
Height first prune	m	1	1	2.6	2.6	1	1
Labour first prune	min $tree^{-1}$	1	1	5.4	5.4	3.1	15
Height last prune	m	8	8	8	8	6	8
Labour last prune	min $tree^{-1}$	15	15	47	28	3.8	48
Removal of prunings	min $tree^{-1}$	4	4	2.4	2.4	2.4	2.4
Harvest cost (total)	€ ha^{-1}	584	584	0	0	258	927
Tree cutting	min $tree^{-1}$	7	7	23	23	6	27

Table C4. Financial equivalent annual value (EAV; € ha^{-1} y^{-1}), the environmental externalities EAV, and the societal EAV as related to the carbon value € (t CO_2)$^{-1}$ for the societal EAV of 1) agroforestry to equalize the arable, and of 2) the tree only system to equalize the arable, per country

Country	Scenario		Financial		E. externalities		Societal	
			With grants	Without grants	With grants	Without grants	With grants	Without grants
United Kingdom	1	Carbon value	-	-	16	16	-	-
		Arable	559	315	-38	-38	**522**	**277**
		Agroforestry	457	212	65	65	**522**	**277**
	2	Carbon value	-	-	64	32	-	-
		Arable	559	315	-156	-79	**403**	**236**
		Tree-only	69	69	334	167	**403**	**236**
Spain	1	Carbon value	-	-	185	137	-	-
		Arable	358	205	-305	-226	**54**	**-21**
		Agroforestry	199	86	-145	-108	**54**	**-21**
	2	Carbon value	-	-	137	109	-	-
		Arable	358	205	-226	-179	**132**	**26**
		Tree-only	48	-41	84	67	**132**	**26**
Switzerland	1	Carbon value	-	-	40	345	-	-
		Arable	2,222	861	-45	-392	**2,177**	**469**
		Agroforestry	1,961	-1,404	216	1,874	**2,177**	**469**
	2	Carbon value	-	-	340	136	-	-
		Arable	2,222	862	-386	-155	**1,836**	**706**
		Tree-only	-48	-48	1,884	754	**1,836**	**706**

Appendix D

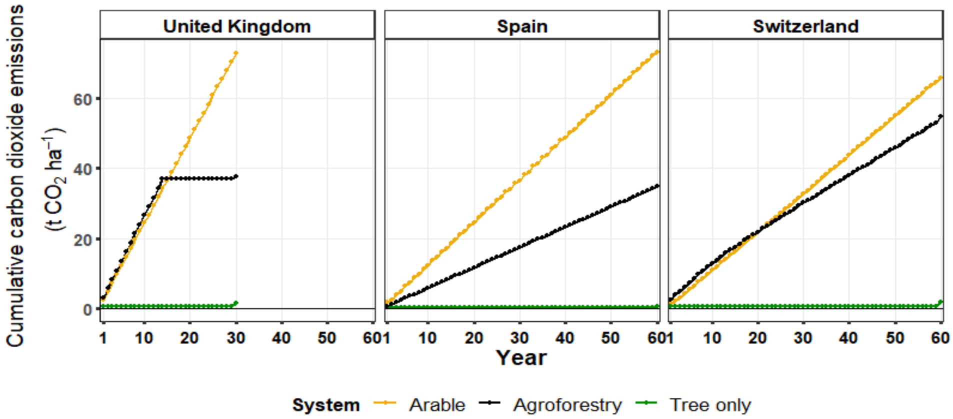

Figure D1. Modelled cumulative CO_2 emissions for the Arable, Agroforestry and Tree-only for UK, Spain and Switzerland over 30, 60 and 60 years respectively

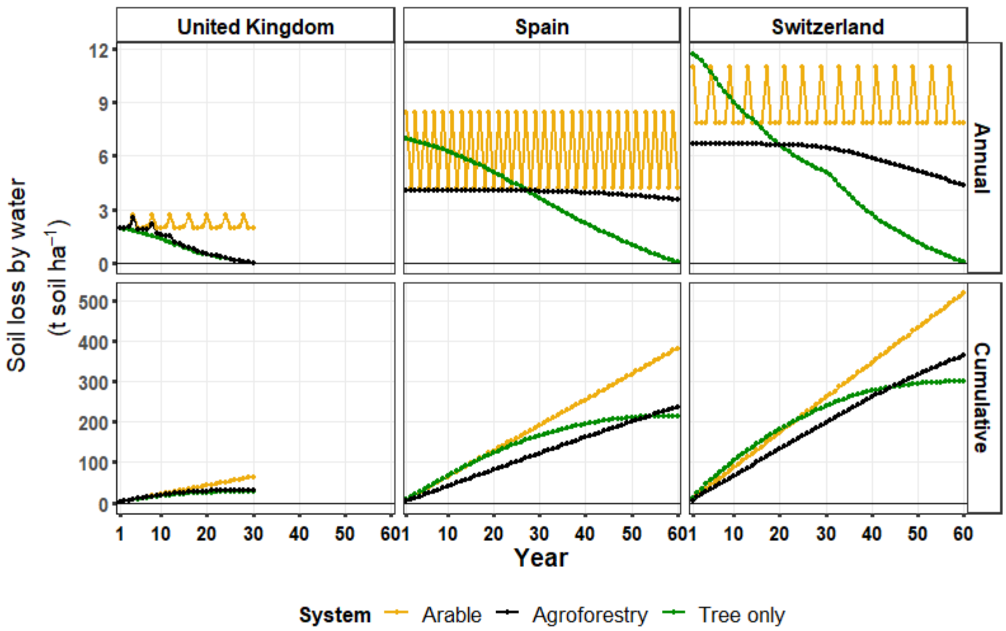

Figure D2. Modelled annual and cumulative soil loss by water (t ha^{-1} y^{-1}) for the Arable, Agroforestry and Tree-only for UK, Spain and Switzerland over 30, 60 and 60 years respectively

Declaration of interests

☒ The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

☐The authors declare the following financial interests/personal relationships which may be considered as potential competing interests:

Whole system valuation of arable, agroforestry and tree-only systems at three case study sites in Europe

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