Bio-economic process-based modelling methodology for measuring and evaluating the ecosystem services provided by agroforestry systems

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Bio-economic process-based modelling methodology for measuring and evaluating the ecosystem services provided by agroforestry systems

Josep Crous Duran, 2022

Abstract

Agroforestry integrates woody vegetation with crop and/or animal production. This combination can benefit from ecological and economic interactions that allow better use of natural resources and improved economic performance. But despite financial support offered through policy, the implementation of new agroforestry systems has not been widespread in the European Union. This thesis aimed to develop additional scientific knowledge on the potential of agroforestry systems in terms of productivity and environmental benefits. The method consisted in improving a bio-physical process-based model (Yield-SAFE) and an integrated bio-economic model (Farm-SAFE) and using both to model four different agroforestry systems in different edaphoclimatic conditions in Europe. Four different agroforestry tree-densities were compared to no-tree and tree-only monoculture alternatives. The results showed that: 1) in terms of productivity, the inclusion of trees in agricultural land increases the overall accumulated energy but the accumulated energy per tree decreases as the tree density of trees increases; 2) agroforestry options present a greater capacity of reducing soil erosion, nitrate leaching and increases the carbon sequestration; 3) agroforestry systems can act as more sustainable methods of food production and 4) options without trees are more interesting financially but these option are also the most polluting. And even though the consideration of a monetary valuation of the environmental services offered, agroforestry options would just become more interesting if there is a change on how public financial help is allocated to the sector. The findings of this work reflect what has been previously seen in scientific literature, particularly in terms of the capacity of agroforestry systems to be more productive than monoculture systems, whilst at the same time providing environmental benefits. However, relatively low profitability means that they still fail to attract farmers, the main agents of agroforestry uptake and currently, arable and forestry tend to receive higher subsidies making these land uses more attractive to farmers but considering environmental benefits would make agroforestry a more interesting option.

Keywords

Yield-SAFE, tree density, Carbon Balance Method, Sustainable Intensification

Bio-economic process-based modelling methodology for measuring and evaluating the ecosystem services provided by agroforestry systems

Josep Crous Duran, 2022

Resumo

Os sistemas agroflorestais integram vegetação lenhosa, produção agrícola e/ou produção agropecuária. Esta combinação se beneficia de interações ecológicas e económicas que permitem que os diversos elementos usem melhor os recursos naturais disponíveis. Apesar do apoio financeiro oferecido pela União Europeia na instalação deste tipo de sistema não tem sido bem-sucedida na. O presente trabalho tem como objetivo principal, seguindo uma abordagem de modelação biofísica e bio económica, fornecer respostas para questões relacionadas com as potencialidades dos sistemas agroflorestais em termos de produtividade, benefícios ambientais e viabilidade financeira e económica. A metodologia usa um modelo biofísico de base processual (Yield-SAFE) com um modelo bio económico compatível (Farm-SAFE) e é aplicado em quatro sistemas agroflorestais na Europa. Quatro diferentes densidades crescentes de árvores foram comparadas às suas principais alternativas de cultivo (sem árvores) e floresta (apenas árvores). Os resultados mostram que: 1) em termos de produtividade a inclusão de árvores em terras agrícolas aumenta a energia total acumulada mas a energia acumulada por árvore reduz-se à medida que a densidade de árvores aumenta; 2) existe uma maior capacidade dos sistemas de reduzir o solo erodido, o lixiviamento de nitratos e aumenta a capacidade dos sistemas de armazenar carbono; 3) os sistemas agroflorestais podem atuar como métodos mas sustentáveis de produção e 4) financeiramente as opções sem árvores são mais interessantes mas também as mais poluentes. Contudo considerando a valorização monetária atual das externalidades as opções agroflorestais só seriam mais interessantes se houvesse uma mudança na distribuição das ajudas públicas atribuídas ao sector. Todos estes resultados são semelhantes aos observados na literatura científica. Atualmente, a agricultura é altamente subsidiada, no entanto, entrar a considerar alguns fatores como os benefícios ambientais poderiam melhorar o interesse financeiro e económico dos sistemas agroflorestais.

Palabras chave

Yield-SAFE, densidade arbórea, Método do Balanço de Carbono, Intensificação sustentável

Extended abstract

During recent years it has been seen that agroforestry practices, defined as being the integration of woody vegetation with crop and/or animal production, benefits, if compared to crop or tree monoculture alternatives, from the ecological and economic interactions associated to the presence of more than one element in the system. But despite these benefits and the financial support offered by the administration, the implementation of this type of systems has not being successful in the European Union. The present work has the main objective to have and in-depth look at the reasons of this low rate of implementation and offer some evidence of the potentialities of agroforestry systems. In this sense the methodology follows a bio-physical an economical modelling approach for the development of an integrated assessment of agroforestry systems as sustainable practices. However, during the first stage of this work, and as a preparatory study, the candidate collaborated in a social survey among relevant stakeholders at a European scale to better identify the positive aspects and the main barriers and constrains associated to the implementation of agroforestry practices. The respondents mostly associated the positive aspects to the additional environmental benefits supplied while the main barriers were mostly related to the unknow productivity performance, the management complexity and the financial viability of these practices. Following these answers, the objectives were established in order to put some light to the identified lack of knowledge. In the first of the exercises, how specific agroforestry designs and tree densities affect productivity was testes by using a process-based bio-physical model, called Yield-SAFE that was updated to quantify the food, material and energy (Provisioning Ecosystem Services) production. The updated version was used in four contrasting agroforestry case study systems and its arable and forestry alternatives in Europe. Results showed that including trees in pasture or arable systems increased the overall accumulated energy of the system in comparison with monoculture forestry, pasture, and arable systems, but that the accumulated energy per tree was reduced as tree density increased. On a second exercise, the capacity of the same systems to supply environmental benefits (Regulating Ecosystem Services) was tested using the same bio-physical modelling approach and by implementing methodologies for estimating

soil erosion, nitrate leaching and carbon sequestration to the Yield-SAFE model. In this case, results confirmed the idea of stakeholders that by including trees on farmland this generally improved the capacity of systems to reduce the soil eroded, the nitrate leached and increased the capacity of the systems to store carbon. This exercise considered the same systems and tree densities and therefore confirmed the capacity of agroforestry systems to enhance the environmental benefits supply of the humanized systems while ensuring the provision of food, materials and energy, and confirming agroforestry as a sustainable practice. However, on an specific exercise, both Provision and Regulation Ecosystem Services were combined in order to develop the Carbon Balance Method. This method allows to assess the potential of systems and different management options to be considered as sustainable intensification practices. The method was tested for wheat production in Portugal comparing production in a crop monoculture and a agroforestry alternative. The methodology behind compares the carbon emissions emitted from the production of a certain unit of food and the carbon sequestered associated to the same amount of food produced over a specific area and time. The results confirmed the capacity of agroforestry option to be considered a sustainable intensification way of food production as maintained wheat yields whilst providing a positive carbon balance (meaning carbon is sequestered) after year 50 onwards. Finally, the final exercise tried to answer the missing information related to the farm profitability and economic interest of agroforestry. The approaches followed by this exercise were two. In a first case the point of view of the farmer was considered using a financial analysis while on a second approach the point of view of the wider society was took into account by including a monetary valuation of the externalities. A bio-economic model called Farm-SAFE was used for three of the four systems analysed previously. Results showed that in terms of farm profitability, arable options were always better generating highest incomes, but also the largest damages for the environment. In economic terms however, agroforestry and tree-only systems became slightly more interesting options for land managers in the UK case study but remained behind in the *dehesa/montado* site in Spain and in the Cherry tree pastures of Switzerland. In the first case the slow tree growth did not allow to compensate the low but present environmental benefits offered by the arable option while in Switzerland public subsidies were far too high to compensate any other option.

Therefore, scientific evidence has been offered on the potential performance, financial viability and the environmental benefits provided by agroforestry systems and the capacity of these systems to be used as Sustainable Intensification practices. All these findings are in line with what has been previously seen in scientific literature of the capacity of agroforestry systems to be more productive while offering environmental benefits. However, for the missing financial/economic attractiveness some factors need to be considered being of special importance the point of view guiding the public subsidies. Currently following a farmer point of view, agriculture is subsidized making arable options more attractive. Instead, if societal benefits were promoted arable options would see a reduction in the income generated and the presence of more environmental benefits would enhance tree-present options making agroforestry more interesting.

Resumo alargado

Os sistemas agroflorestais integram vegetação lenhosa, produção agrícola e / ou produção agropecuária. Esta combinação beneficia de interações ecológicas e económicas que permitem que os diferentes elementos presentes usem melhor os recursos naturais disponíveis. Apesar do apoio financeiro oferecido pela administração, a nova instalação deste tipo de sistemas não tem sido bem-sucedida na União Europeia. O presente trabalho tem como objetivo principal, seguindo uma abordagem de modelação biofísica e bio económica, fornecer respostas para questões relacionadas com as potencialidades dos sistemas agroflorestais em termos de produtividade, benefícios ambientais e viabilidade financeira e económica. A metodologia usa um modelo biofísico de base processual (Yield-SAFE) com um modelo bio económico compatível (Farm-SAFE) e é aplicado em quatro sistemas agroflorestais diferentes na Europa. Quatro densidades crescentes de árvores são comparadas às suas principais alternativas de cultivo (sem árvores) e floresta (apenas árvores). No entanto, durante a primeira etapa deste trabalho, o candidato colaborou num estudo mais do tipo social e à escala europeia para identificar os aspetos positivos e as principais barreiras e restrições associadas à implementação de práticas agroflorestais. Os entrevistados associaram principalmente os aspetos positivos aos benefícios ambientais adicionais fornecidos. Enquanto as principais barreiras estavam relacionadas principalmente com o desconhecimento sobre a produtividade destes sistemas e à viabilidade financeira dessas práticas. Seguindo essas respostas, os objetivos foram estabelecidos, sendo o objetivo principal o de melhorar a falta de conhecimento que foi identificado. No primeiro dos exercícios, e para esclarecer como os sistemas agroflorestais e a densidade das árvores podem afetar a produtividade, o modelo Yield-SAFE foi atualizado para poder quantificar os alimentos, materiais e energia fornecidos e poder comparar o desempenho dos sistemas agroflorestais com alternativas sem árvores e apenas florestais. A versão atualizada foi usada em quatro sistemas agroflorestais diferentes na Europa, mas a ferramenta foi projetada para ser facilmente adaptada a outros sistemas e regiões. Os resultados mostraram que a inclusão de árvores em pastagens ou sistemas aráveis aumentou a energia total acumulada no sistema em comparação com monoculturas ou alternativas florestais;

no a energia acumulada por árvore reduz-se à medida que a densidade das árvores aumenta. Num segundo exercício, foi testada a capacidade dos mesmos sistemas de fornecer benefícios ambientais usando a mesma abordagem de modelagem biofísica e implementando metodologias no modelo Yield-SAFE para estimar a erosão do solo, a lixiviação de nitratos e o sequestro de carbono. Nesse caso, os resultados confirmaram a ideia das partes interessadas de que, ao incluir árvores em terras agrícolas, isso geralmente melhorava a capacidade dos sistemas de reduzir o solo erodido, o nitrato lixiviado e aumentava a capacidade dos sistemas de armazenar carbono. Este exercício considerou os mesmos sistemas e densidades arbóreas que o exercício anterior e, portanto, confirmou a capacidade dos sistemas agroflorestais de melhorar o fornecimento de benefícios ambientais garantindo o fornecimento de alimentos, materiais e energia, ou seja considerar os sistemas agroflorestais como práticas de intensificação sustentáveis. A partir destes resultados foi desenvolvido um Método de Balanço de Carbono. Esse método permite avaliar o potencial dos tipos de cultivos como práticas de intensificação sustentável. O método foi testado para a produção de trigo em Portugal, comparando a produção em sistema de monocultivo e um sistema agroflorestal. O método compara as emissões de carbono e o carbono sequestrado em biomassa e solo associados à produção de uma quantidade de alimento numa unidade de área e tempo. Esta unidade permite comparar facilmente os diferentes métodos de produção. Os resultados confirmaram a capacidade dos sistemas agroflorestais de permitirem uma intensificação sustentável de produção em comparação com um monocultivo, pois mantêm a produção de trigo, proporcionando um balanço positivo de carbono (o que significa que o carbono é sequestrado) a partir do ano 50. Finalmente, o exercício final tentou responder às informações relacionadas com a viabilidade financeira e económica dos sistemas agroflorestais. Para este exercício, foram seguidas duas abordagens diferentes: do ponto de vista do agricultor e do ponto de vista da sociedade em geral. Para considerar o ponto de vista do agricultor foi usada uma análise financeira, enquanto na segunda abordagem foi considerada uma avaliação monetária das externalidades. Um modelo bio-económico chamado Farm-SAFE foi usado para três dos quatro sistemas analisados anteriormente. Os resultados mostraram que, em termos de rentabilidade agrícola, as opções aráveis geravam sempre melhores rendimentos, mas essas opções também foram as mais prejudiciais para o ambiente. No entanto, em termos económicos, os sistemas agroflorestais tornaram-se opções ligeiramente mais interessantes para os gestores, mas só conseguiram superar as alternativas aráveis no Reino Undo. No sistema de *dehesa / montado* em Espanha o crescimento lento das árvores não permitiu superar os baixos benefícios ambientais oferecidos pela opção arável, enquanto nos sistemas silvo-pastoris da Suíça, os subsídios públicos são muito altos para compensar qualquer outra opção.

Resumindo, este trabalho conseguiu apresentar algumas evidências sobre o desempenho potencial e os benefícios ambientais proporcionados pelos sistemas agroflorestais e a capacidade desses sistemas para serem utilizados como práticas de intensificação sustentável, ou seja, produzir a mesma ou mais quantidade de um produto, na mesma área e no mesmo tempo com menos impacto ambiental associado. Estes resultados são semelhantes aos observados na literatura científica. No entanto, alguns fatores precisam de ser reconsiderados para melhorar o interesse financeiro e económico destes sistemas. Atualmente, a agricultura é altamente subsidiada, o que faz das opções aráveis as mais interessantes. Alternativamente, se os benefícios sociais fossem considerados e promovidos, as opções com presença de árvores aumentariam a renda gerada tornando os sistemas agroflorestais práticas mais interessantes.

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.

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Table 1. List of national (Portuguese) and international projects that offered financial support to the development of this thesis.

Project	Acronym	Duration	Reference
AGroFORestry that Will Advance Rural Development	AGFORWARD	2013-2017	EU FP7 GA Nº. 613520
Models and Decision Support tools for integrated Forest policy development under global change and associated Risk and Uncertainty	SUFORUN	2016-2020	H2020-MSCA-RISE-2015 GA No. 691149
European non-wood forest products network	NWFPs	2013-2017	COST FP1203-MoU
Unit Project Reference for basic funding	CEF 2019	2019	UID/AGR/00239/2019
Unit Project Reference for basic funding	CEF 2020	2020	UIDB/00239/2020

Preamble

This thesis is structured in eight chapters. **Chapter 1** corresponds to the introductory chapter and presents the study background, exposes the research motivation and lists the main objectives of this study while **Chapter 8** includes the final conclusions, important remarks and proposes future prespectives. The main body of this thesis is presented from **Chapter 2 to Chapter 6.** Each chapter corresponds to a scientific article with the candidate being the main author (**Chapter 3, 4 and 5**) or having played an important role as a co-author by leading the Portuguese case study and helping with the analysis of the results and the writing of the scientific publication (**Chapter 2**) or by helping with the development of the methodology, the finding of the results and the final writing of the derived publication (**Chapter 6**). On its turn, **Chapter 7** collects the abstracts of another 13 scientific publications published that are related to main subject of this thesis and where the candidate has acted as a co-author. The list of the chapters and the correspondent scientific publications are shown in **Table 2**.

Table 2. List of scientific publications and the related chapters.

Chapter	Title
1	General Introduction
2	García de Jalón S et al (2017). How is agroforestry perceived in Europe? An assessment of positive and negative aspects by stakeholders. Agroforestry Systems. DOI: 10.1007/s10457-017-0116-3
3	**Crous-Duran J** et al (2018). Modelling tree density effects on provisioning ecosystem services in Europe. Agroforestry Systems. 4. DOI: 10.1007/s10457-018-0297-4
4	**Crous-Duran J** et al (2020). Quantifying regulating ecosystem services in increasing tree densities in European farmland. Sustainability 2020, 12, 6676; doi:10.3390/su12166676
5	**Crous-Duran J** et al (2019). Assessing food sustainable intensification potential of agroforestry using a carbon balance method. IForest. 12: 85–91. DOI: 10.3832/ifor2578-011
6	Giannitsopoulos ML et al (2020). Whole system valuation of arable, agroforestry and tree-only systems at three case study sites in Europe. Journal of Cleaner Production 269C, 122283. DOI: 10.1016/j.jclepro.2020.122283
7	Other scientific publications
8	Conclusions

Communication activities

Scientific publications

2017

den Herder M, Moreno G, Mosquera-Losada RM, Palma JHN, Sidiropoulou A, Santiago Freijanes JJ, **Crous-Duran J,** Paulo JA, Tomé M, Pantera A, Papanastasis VP, Mantzanas K, Pachana P, Papadopoulos A, Plieninger T, Burgess PJ. Current extent and stratification of agroforestry in the European Union. Agriculture, Ecosystems & Environment 241, 121-132. DOI: 10.1016/j.agee.2017.03.005

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Kay S, **Crous-Duran J**, Ferreiro-Domínguez N, García de Jalón S, Graves A, Moreno G, Mosquera-Losada MR, Palma JHN, Roces-Díaz JV, Santiago-Freijanes JJ, Szerencsits E, Weibel R, Herzog F (2017). Spatial similarities between European agroforestry systems and ecosystem services at the landscape scale. Agroforestry Systems. DOI: 10.1007/s10457-017-0132-3

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2018

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García de Jalón S, Graves A, Moreno G, Palma JHN, **Crous-Duran J**, Kay S, Burgess PJ (2018). Forage-SAFE: a model for assessing the impact of tree cover on wood pasture profitability. Ecological Modelling. 372: 24–32. DOI: 10.1016/j.ecolmodel.2018.01.017

2019

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(Oral presentation) **Crous-Duran J**, Graves AR, Kay S, García de Jalón S, Paulo J, Tomé M, Palma JHN. Using a process-based model to assess tree density effects on the supply of regulating ecosystem services. 9th Ecosystem Services Partnership (ESP) World Conference. 11-15 December 2017, Shenzen, China.

(Oral Presentation) García De Jalón S, Graves AR, Moreno G, Palma JHN, **Crous-Duran J**, Oliveira TS and Burgess PJ. Forage-SAFE: a tool to assess the management and economics of wood pasture systems. 15th International Conference on Environmental Science and Technology, Rhodes, Greece.

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(Oral presentation) Paulo, JA and **Crous-Duran, J**. Investigação e transferência de conhecimento para a promoção de modelos de gestão agroflorestais sustentáveis. 'Floresta Global' Seminar. Agroglobal. 7th September 2018.

(Poster) Kay S, Roces-Díaz J, **Crous-Duran J**, Giannitsopoulos M, Graves A, den Herder M, Moreno G, Mosquera-Losada MR, Pantera A, Palma J, Szerencsits E, Herzog F. Agroforestry can mitigate environmental problems in European agricultural deficit areas, IV European Agroforestry Conference, 28-30 May 2018, Nijmengen, The Netherlands.

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(Oral presentation) Kay S, Graves AR, Palma JHN, Moreno G, Roces-Díaz J, **Crous-Duran J**, Aviron S, Chouvardas D, Mosquera-Losada R, Szerencsits E, Torralba M, Herzog F. Benefits of temperate agroforestry landscapes - economic evaluation of the marketable and the non-marketable outcomes. 4^{th} World Congress on Agroforestry, 20-22 May, Montpellier, France.

(Oral presentation) Gidey Bezabeh T, Palma JHN, Oliveira TS, **Crous-Duran J.** Using Yield-SAFE model to assess impacts of climate change on yield of Coffee under agroforestry and monoculture systems. 4^{th} World Congress on Agroforestry, 20-22 May, Montpellier, France.

(Oral presentation) Palma JHN, Graves AR, **Crous Duran J**, García de Jalón S, Oliveira TS, Paulo JA, Ferreiro-Domínguez N, Moreno G. EcoYield-SAFE: maintaining a parameter-sparse approach in modelling ecosystems processes and dynamics. 4^{th} World Congress on Agroforestry, 20-22 May, Montpellier, France.

(Poster) Kay S, Roces-Diaz J, **Crous-Duran J**, Giannitsopoulos M, Graves AR, den Herder M, Moreno G, Mosquera-Losada R, Pantera A, Palma JHN, Paracchini M, Rega C, Rolo V, Rosati A, Smith J, Szerencsits E. New agroforestry on European ecosystem service deficit farmland can compensate up to 43% of agricultural GHG emissions. 4^{th} World Congress on Agroforestry, 20-22 May, Montpellier, France.

Doctoral curricular courses

The SUSFOR doctoral programme includes the presentation of an original research work presented in the form of a dissertation or thesis (the present document) but also a complementary curricular part with a curricular charge of 30 credits of the European Credits Transfer and Accumulation System (ECTS). Of these 30 ECTS, 15 ECTS were mandatory while the other 15 ECTS were selected by the doctoral student. **Table 3** shows the mandatory and optional courses the candidate undertook during the doctoral program and the correspondent evaluation (marks) received.

Table 3. Curricular courses and marks of the courses attended by the candidate.

Mandatory	**Institution**	**ECTS**	**Marks**
Scientific Motivation Seminar	ISA	3	16/20
Research Project Presentation week	ISA	6	18/20
Advanced course on Statistical and mathematical tools for ecological data analysis	ISA	6	17/20
Optional	**Institution**	**ECTS**	**Marks**
Economia do Ambiente	ISA	6	15/20
Modelaçao em GIS	UNova	7,5	16/20
An introduction to interactive programming in python	Online/Rice University	1,5	NA

Other courses

During the doctoral program the candidate also attended additional short courses in order to improve the skills in scientific writing and process-based modelling. **Table 4** shows the details of these courses.

Table 4. Additional courses attended by the doctoral student.

Additional courses	**Institution**	**Hours**	**Evaluation**
Scientific writing and publication. Research paper and thesis.	IST	21	NA
Agroforestry Modelling with Hi-SAFE	INRA	8	NA

Research Missions

The candidate also joined three different research missions. These missions were funded under the COST Action FP1203 and the SUFORUN project and allowed the candidate to stay for a short term (between 10 days and 5 months) in other institution and research centres. The objective of this type of missions was to promote knowledge exchange and collaboration between research institutions. The details of the missions undertaken by the candidate are shown in **Table 5**.

Table 5. Research missions undertaken by the doctoral student.

Research missions	**Institution**	**Duration**	**Funding**
Development of a methodological approach for evaluating provision of ecosystem services from cork oak agroforestry systems	BTU, Cottbus (Germany)	12/01-23/01/2015	Cost Action: FP-1203
Modelling holm oak acorn production in southern Spain	UEX Plasencia (Spain).	18/05-29/05/ 2015	Cost Action: FP-1203
Calibration of the Yield-SAFE model for coffee agroforestry systems in Costa Rica	CATIE, Turrialba (Costa Rica)	11/01-18/05/2018	SUFORUN exchange project

Chapter 1| Introduction

General introduction

After the Convention on Climate Change in Rio de Janeiro, in 1992, the concept of Ecosystem Services (ES) was discussed and several definitions were proposed (Daily 1997; Costanza et al. 1997; Millennium Ecosystem Assessment et al. 2005). In Fisher et al (2009), ES are seen as aspects of ecosystems that actively or passively were used to provide human well-being. After several attempts of classification (Wallace 2007), ES were finally divided into three main categories directly affecting people: 1) Provisioning services which consider food, materials, or energy outputs from ecosystems; 2) Regulating services which include services where ecosystems act as regulators on water, soil, or air quality; and 3) Cultural services including aspects related to the recreation and subjective services offered by ecosystems. A fourth category was also considered: Supporting services which included those facilitating the presence of the other three categories (Millennium Ecosystem Assessment et al. 2005).

A consensus is also growing on classifying these contributions as intermediate or final services. Intermediate services are ecosystem characteristics measured as ecosystem structure, processes, and functions that support final services. Final services are components of nature possessing an explicit connection to human well-being, meaning that they have direct value to society (Boyd and Banzhaf 2007; Fisher et al. 2009). Therefore, the amount of human welfare provided depends on the ecological conditions of the respective ecosystems which, in turn, are affected by how they are managed (Lima-Santos, personal communication).

In order for land-managers and other decision-makers to be able to put into practice this ES concept, credible and legitimate measurements are needed to estimate the potential existing trade-offs between ES (Maes et al. 2012). In ecology, bio-physical models (empirical or process-based) are usually used to estimate how specific ecosystem indicators evolve at different spatial and temporal scales. However, a consensus is emerging in the scientific community that the impacts of management decisions are best made based on process-based models (Cuddington et al. 2013; Wong et al. 2014). This seems to be the most promising option for bio-physical quantification of the existing relationships within ecosystems, for estimating how these systems are modified by human management, and for quantifying the final services they provide (Cuddington et al 2013).

But there is also a need for management decisions over ecosystems to meet the rising level of demand required due to global population growth. This growth has specially increased the demand for Provisioning services i.e. food, materials and energy, adding greater pressure on the environment and reinforced the negative impacts associated with their production, such as GHG emissions, soil loss, water pollution or land use conversion.

In 2009 the Food and Agriculture Organization of the United Nations (FAO 2009) stated that the challenges associated with food production should be tackled on the same land parcel by considering Sustainable Intensification (SI) practices. These practices are defined as those that are able to ensure food security while maintaining biodiversity and ecosystems services (Godfray et al. 2012; Godfray and Garnett 2014).

Agroforestry practices are already considered one of these SI practices as the integration of woody vegetation with crop and/or animal production benefits from ecological and economic interactions. These interactions allow a multifunctional land use that combines food, energy, and material provision with an environmental improvement and a reinforcement of local economies (Jose 2009).

Some agroforestry systems, such as wood pastures, have been practiced in Europe since Neolithic times. During the 20th century, mechanization and intensive land use management practices have led to an increased separation between farming and tree management and, consequently, agroforestry practices were restricted to marginal and/or degraded areas. However, agroforestry systems are still particularly common in the Mediterranean basin where they are valued for their multifunctional capacity and contribution to local economies (Mosquera-Losada et al. 2012).

More recently, scientific research has been stressing the benefits agroforestry systems can offer for society. Additionally, it has provided scientific knowledge and tools to support decision-makers managing synergies and trade-offs between production and other ES in multi-functional working landscapes. It has been acknowledged that agroforestry practices can: improve the efficiency in resource use (light, water, soil, nutrients); improve edapho-climatic conditions within a system (reduction of wind speed, temperature buffering, and soil moisture); help to mitigate soil erosion and nitrate leaching problems; enhance landscape biodiversity; lead to an overall higher biomass production for material or energy conversion (fuelwood); and thus match the increasing demand for bio-energy self-supply in rural decentralized areas (Palma et al., 2007a; Jose, 2009).

Research motivation and problem definition

Promising sustainable benefits provided by agroforestry practices prompted the European Union (EU) to promote the establishment of these systems by means of the Common Agriculture Policy (CAP). For the programme period 2007-2013 and as part of Pillar II: improving the environment and the countryside, measure 222 promoted the establishment of agroforestry systems in the EU. However, results were rather poor and only 6.4% of the allocated budget for all the EU was finally used (Pisanelli et al., 2014), leading to just around 3,000 hectares of new agroforestry systems (Hodosi and Szedlak 2018) when, at that time, an estimated 40% of agricultural land in Europe could be targeted to implement an agroforestry system that could mitigate environmental problems (Reisner et al. 2007). In the current CAP (2014-2020), agroforestry received further support through Pillar II and Article 23 of Regulation 1305/2013 which provides the possibility of establishing and supporting the regeneration or renovation of existing agroforestry systems under Measure 8.2. This measure covers the establishment and up to 5 years of maintenance costs. However, the measure was optional for Member State (MS) and was just implemented by one every five regions in the EU. But while final results are not published, consulted experts suggest that the final agroforestry area implemented will be far less than the 74.000 hectares foreseen.

Although the financial support provided by the public administration was supposed to increase adoption of agroforestry systems, farm-level decisions are ultimately made by producers, landowners or by other key stakeholders with relevant influence. Despite research that has been able to highlight the environmental, social and economic benefits for rural areas associated with these practices and even though farmers are already aware of the benefits agroforestry could bring to their areas (Graves et al. 2009; 2017), studies have also shown that the implementation of agroforestry systems can lead to a loss of farm income, reduced labour productivity and an increase in complexity of work (Pannell 1999; Graves et al. 2009; Burgess et al. 2016; Graves et al. 2017). This highlights the importance of better farm management and planning decisions as these then become even more critical in determining the economic performance of the system (Schroth et al. 2001). In this sense, a consistent understanding of the perception these stakeholders have for agroforestry, including the negative and positive aspects, is needed as a first step for the design and development of practical tools able to help during decision-making to support the implementation of appropriate policy measures for increasing the adoption rates of these agroforestry practices in Europe.

During the AGFORWARD project, a four-year research project funded by the EU with the purpose of promoting agroforestry practices in Europe (Burgess and Rosati 2018), 40 stakeholder groups across Europe were established and meetings amongst them organized in order to discuss and identify key opportunities and constraints related to agroforestry practices and management. The meetings confirmed that for stakeholders, positive aspects were related to production levels and environment benefits while negative aspects were mostly associated with management and socio-economic issues (Burgess 2014; Crous-Duran et al. 2014; Tsonkova and Mirck 2014).

Objectives and outline

The concerns identified in the stakeholder meetings helped to define the objectives of this work. In this sense, the main objective of this thesis was to generate robust information using a bio-physical and bio-economic process-based modelling approach to consider different management options - such as tree density or the thinning regimes. The approach developed would allow the accomplishment of the following objectives:

1) To quantify the growth and yields and the final Provisioning Ecosystem Services provided (food, materials and energy) of agroforestry systems in

Europe and compare this performance to related arable/pastures monoculture or forestry alternatives.

2) To quantify the capacity of the agroforestry systems to offer environmental benefits (Regulating Ecosystem Services) and compare this to related arable/pasture monoculture or forestry alternatives in Europe.
3) To analyse the potential of agroforestry systems to be recognised as sustainable food intensification practices by assessing the supply of Provisioning and Regulating Ecosystem Services from the same area of land.
4) To evaluate the financial and broader economic performance of agroforestry systems in comparison to the arable/pasture monocultures and forestry alternatives through the quantification and valuation of the environmental externalities associated with each of the systems.

The work is been presented in eight different chapters.

Chapter 1 (this chapter) presents a general introduction and describes the main objectives and the structure of this research.

Chapter 2 presents a journal paper examining social perception of agroforestry by stakeholders that led to the objectives identified for this thesis**.** This chapter describes a pan-European analysis of how key actors including farmers, landowners, agricultural advisors, researchers and environmentalists perceive agroforestry in Europe. With 344 valid responses from 11 different European countries, the study confirms that agroforestry is seen as a practice enhancing the environmental value of agriculture and the landscape but its complexity of work, the increased labour needed, and the lack of knowledge on the management practices and financial viability of the systems act as barriers to implementation of the systems.

Chapter 3 assesses the productivity of the agroforestry systems and compares the performances of these practices to arable, pasture, and forestry monoculture alternatives. With this objective, the bio-physical Yield-SAFE model was adapted to quantify the food, material, and biomass energy production of four contrasting case study systems in Europe and then the Provisioning Ecosystem Services were analysed for different tree density ranging from a no-tree alternative to a forestry alternative. The results were translated into a common energy unit in order to facilitate the comparison and showed firstly, that by including trees in pasture or arable systems the overall accumulated energy of the system increased compared to monoculture forestry, pasture, or arable systems, and secondly, that the additional energy accumulated per tree was reduced as tree density increased.

Chapter 4 evaluates the capacity of agroforestry practices to improve the environmental benefits. In this sense, the Yield-SAFE model was updated with methodologies for quantifying the amount of soil eroded, the nitrate leached, and the carbon sequestered for the same agroforestry systems with the same management alternatives as in Chapter 3, were modelled in order to evaluate their capacity to supply Regulating ES. Results showed a consistent improvement of the supply of these services can be expected when introducing trees in the farming landscapes in different environmental regions in Europe. Even though the forestry alternatives provided the most RES, from an arable or pure pasture alternative starting point, the addition of trees provided a reduction in soil erosion and nitrates leached, and an increase in carbon sequestration.

In **Chapter 5** both Provisioning ES (Chapter 3) and Regulating ES (Chapter 4) are used to assess the potential of different management options as sustainable food intensification practices, where increasing food production per area of land is obtained whilst reducing the associated environmental impacts. The method is based on comparing the carbon emissions and the carbon sequestered by different systems when producing a given quantity of food produced over a specific area and over a specific time-frame. This method was tested in Portugal by comparing wheat production under a crop monoculture and agroforestry systems. The results showed that the agroforestry systems were a suitable practice for sustainable intensification compared to the crop monoculture as it provided wheat whilst providing a positive carbon balance from year 50 onwards.

Chapter 6 consists of a study that evaluates both financial profitability for land-managers and the benefits for society of arable, agroforestry and tree-only alternatives in three different regions in Europe. For this study the bio-physical modelling undertaken with Yield-SAFE was combined with a bio-economic model called Farm-SAFE in order to compare the financial (EAV_F) and the economic (or societal) (EAV_E) equivalent annual values by including monetary values for five environmental externalities. Across the three case studies, arable farming generated higher farm incomes than the agroforestry or tree-only systems, but also created the greatest environmental costs. However, at the current externalities prices considered, the EAV_E of the agroforestry and tree-only systems were greater or similar to that of the arable system only in the UK.

Chapter 7 collects the abstracts of up to another 13 scientific articles published where the candidate has been involved as co-author and that have not been included in the main body of this thesis but that have a strong link to the main subject i.e. the Ecosystem Services provided by agroforestry systems. These publications are related to the social-cultural services that are associated with agroforestry systems (3); the development of methods and improvement of models for bio-physical growth estimation of agroforestry systems (1); bio-economic performance (2); the analysis of the ecosystem services provided by agroforestry systems at a landscape level (4); the estimation of the current distribution of these systems in Europe (1) and the use of the Yield-SAFE model for agroforestry systems outside Europe (1).

Finally, **Chapter 8** synthesises and summarizes the research providing a general conclusion and recommendations for future research.

Chapter 2| How is agroforestry perceived in Europe? An assessment of positive and negative aspects by stakeholders

Based on García de Jalón, S., Burgess, P.J., Graves, A., Moreno, G., McAdam, J., Pottier, E., Novak, S., Bondesan, V., Mosquera-Losada, R., **Crous-Duran, J**., Palma, J.H.N., Paulo, J.A., Oliveira, T.S., Cirou, E., Hannachi, Y., Pantera, A., Wartelle, R., Kay, S., Malignier, N., van Lerberghe, P., Tsonkova, P., Mirck, J., Rois, M., Kongsted, A.G., Thenail, C., Luske, B., Berg, S., Gosme, M., Vityi, A., 2017. How is agroforestry perceived in Europe? An assessment of positive and negative aspects by stakeholders. Agrofor. Syst. https://doi.org/10.1007/s10457-017-0116-3

Abstract

Whilst the benefits of agroforestry are widely recognised in tropical latitudes few studies have assessed how agroforestry is perceived in temperate latitudes. This study evaluates how stakeholders and key actors including farmers, landowners, agricultural advisors, researchers and environmentalists perceive the implementation and expansion of agroforestry in Europe. Meetings were held with 30 stakeholder groups covering different agroforestry systems in 2014 in eleven EU countries (Denmark, France, Germany, Greece, Hungary, Italy, Netherlands, Portugal, Spain, Sweden and the United Kingdom). In total 344 valid responses were received to a questionnaire where stakeholders were asked to rank the positive and negative aspects of implementing agroforestry in their region. Improved biodiversity and wildlife habitats, animal health and welfare, and landscape aesthetics were seen as the main positive aspects of agroforestry. By contrast, increased labour, complexity of work, management costs and administrative burden were seen as the most important negative aspects. Overall, improving the environmental value of agriculture was seen as the main benefit of agroforestry, whilst management and socio-economic issues were seen as the greatest barriers. The great variability in the opportunities and barriers of the systems suggests enhanced adoption of agroforestry across Europe will be most likely to occur with specific initiatives for each type of system.

Keywords

Agroforestry, Adoption, Barrier, Opportunity, Europe

Introduction

From the 1960s to the beginning of the twenty-first century, crop yields per unit area in Europe have increased as a result of plant breeding, the use of external inputs such as fertilizers and pesticides, and the use of specialised field machinery (Burgess and Morris 2009). This change from traditional to modern agricultural systems has led to a simplification and standardisation of farming systems and to a substantial loss of landscape heterogeneity (Dupraz et al. 2005). At the same time, the area occupied by traditional agroforestry practices (mainly associated with the integration of trees and farming) has declined across Europe. However, agroforestry is still practised on 15.4 million hectares in Europe, about 3.6% of the total territorial area of the European Union (EU) (den Herder et al. 2017).

FAO (2015) defines agroforestry as “land-use systems and technologies where woody perennials (trees, shrubs, palms, bamboos, etc.) are deliberately used on the same land-management units as agricultural crops and/or animals, in some form of spatial arrangement or temporal sequence”. The two main types of agroforestry on agricultural land are: i) silvo-pastoral systems that typically integrate trees with pasture and domesticated animals and ii) silvo-arable (or agro-silvo-cultural) systems that integrate trees and crops. The combination of trees, animals and arable crops are sometimes referred to as agrosilvo-pastoral systems. In Europe, the AGFORWARD project identified four different categories of agroforestry in terms of the main focus of production and management (Burgess et al. 2015): i) agroforestry of high nature and cultural value (e.g. traditional systems such as the *dehesa, montado* and other forms of wood pasture and hedgerows which are widely recognised for their biodiversity and heritage), ii) agroforestry with high value trees (e.g. grazed or intercropped orchards or olive groves where tree crops is the primary focus), iii) agroforestry for arable farmers where the crop component is the main focus of the production (e.g. tree lines and windbreaks in arable systems), and iv) agroforestry for livestock farmers, when livestock is the main focus (e.g. fodder trees for ruminants or hens in woodlands).

In 2005, the establishment of agroforestry on agricultural land was supported by the EU Regulation 1698/2005, and the “high ecological and social value” of agroforestry was recognised. Although this support was supposed to increase adoption, farm-level decisions are ultimately made by producers, landowners or by other key stakeholders with relevant influence. Thus, a better understanding of stakeholders' perception of agroforestry is essential to design appropriate policy measures and tools.

Research has highlighted multiple benefits of agroforestry in Europe in terms of environmental benefits (e.g. ecological values and biodiversity), social benefits (e.g. rural employment and cultural practices) and economic benefits (e.g. diversified source of income) (Eichhorn et al. 2006; Plieninger et al. 2015; Fagerholm et al. 2016). However, agroforestry has also been associated with a loss in farm income, reduced labour productivity, and an increase in complexity of work (Pannell 1999; Graves et al. 2009; Burgess et al. 2016; Graves et al. 2017). The latter means that farm management and planning decisions become more critical in determining the economic performance of the system (Schroth et al. 2001). For example, the introduction of trees into arable fields, whilst providing an additional source of future revenue in the form of timber, also shades the crop and alters its capture and use of soil water and nutrients (Schroth et al. 2001). Whilst the crop-tree interaction, if managed correctly, can improve the economic performance of the farm the system does become more complex. Consequently, agroforestry farmers need to consider more variables in their decision-making process including temporal and spatial factors. These, for example include decisions on the orientation of tree rows, the width of the rows, the timing of field operations, and the potential to damage the tree or crop when implementing field operations. Thus farmers' views on how they could deal with these agroforestry operations and how agroforestry would perform in economic terms on their farms, is likely to determine adoption.

Various studies have assessed farmer attitudes towards conservation practices (e.g. Barnes et al., 2009; Howley et al., 2014; Reimer et al., 2012). However, there are not many specifically focused on agroforestry, and in most cases, they refer to case studies in tropical climates (e.g. Babu and Rajasekaran, 1991; Jerneck and Olsson, 2013; Meijer et al., 2015). In Europe, the number of studies assessing farmer attitudes towards agroforestry is relatively small. Graves et al. (2009) analysed farmer perceptions of silvo-arable systems in seven European countries. The study found that whilst in

Mediterranean areas, farmers tended to feel that the principal benefit of silvo-arable systems would be increased farm profitability, in Northern Europe farmers placed greatest value on environmental benefits. By contrast, when asked to identify the greatest negative attribute, Mediterranean farmers identified intercrop yield decline, whereas farmers in Northern Europe highlighted the general complexity of work and difficulties with mechanisation. Liagre et al. (2005) found that the majority of European farmers did not know who had planted the existing isolated trees on their farm and stated that they were present when they started to farm. They also showed that a number of farmers recognised that they often cut the trees without replacement as the trees age and only a small percentage of farmers had planted trees on their farm. Graves et al. (2017) evaluated farmers' views on the benefits, constraints, and opportunities for silvo-arable systems in Bedfordshire, England. The study showed that most farmers felt that silvo-arable systems would not be profitable on their farms and that benefits would tend to be environmental or social rather than financial. The study concluded that management and use of machinery is an important barrier to the adoption of silvo-arable systems.

Using the framework used by Botha and Coutss (2011), the implementation of agroforestry depends on the motivation to change and the capacity to change. The motivation to change is dependent on the removal of barriers to adoption of new systems and the generation of, or existence of, capacity to execute that change. The main objective of this study is to assess how stakeholders and key actors perceive the positive and negative issues of implementing agroforestry practices in Europe and to explore possible methods for promoting agroforestry. The study presents the results of a survey carried out across Europe to analyse how stakeholders perceived the positive and negative aspects of implementing and expanding different agroforestry systems. This work assesses farmer attitudes towards agroforestry in Europe in line with previous studies (e.g. Graves et al., 2009, 2017; Liagre et al., 2005) but advances this by separately assessing the positive and negative aspects for each type of agroforestry system and making comparisons across Europe.

Materials & Methods

Data collection

Data were obtained from a survey and focus group discussions carried out in case-study workshops in Europe with stakeholders and key actors between June and December 2014. The survey was sent and/or handed out in 45 case-study workshops. Of these, participants in 30 of the workshops successfully completed the study, in six workshops the responses did not provide the disaggregated data necessary to make case-study comparisons and in nine workshops the survey was not undertaken. Each case-study workshop represented a different type of agroforestry system located in eleven countries (Denmark, France, Germany, Greece, Hungary, Italy, Netherlands, Portugal, Spain, Sweden and the United Kingdom). **Table 6** describes the 30 case-study workshops used in this study and **Figure 1** shows the geographical location.

In each case-study workshop, a focus group discussion was used to gather information on the barriers and opportunities of implementing and expanding a specific agroforestry system that was pertinent to the local region. Subsequently, a questionnaire was handed to each participant. In the questionnaire, stakeholders were asked to identify and rank the main positive and negative aspects of agroforestry in terms of production, environmental, management, and socio-economic aspects. A total of 45 aspects were evaluated (**Table 7**). Whilst the workshops were primarily focused on qualitative questions, the questionnaire was used to provide a quantitative estimate of the positive and negative attributes of agroforestry. The qualitative data collected in the workshops were used to better explain the survey results. Among the 30 workshops, 344 surveys were successfully completed and returned as presented in **Table 6**.

Workshop participants included producers, landowners, agricultural advisors, members of NGOs, and researchers. Although most participants were local farmers with some experience in agroforestry practices the proportion of stakeholder groups varied in each case study (**Table 6**). Further information for each case-study workshop is presented in reports available on the AGFORWARD project website (www.agforward.eu).

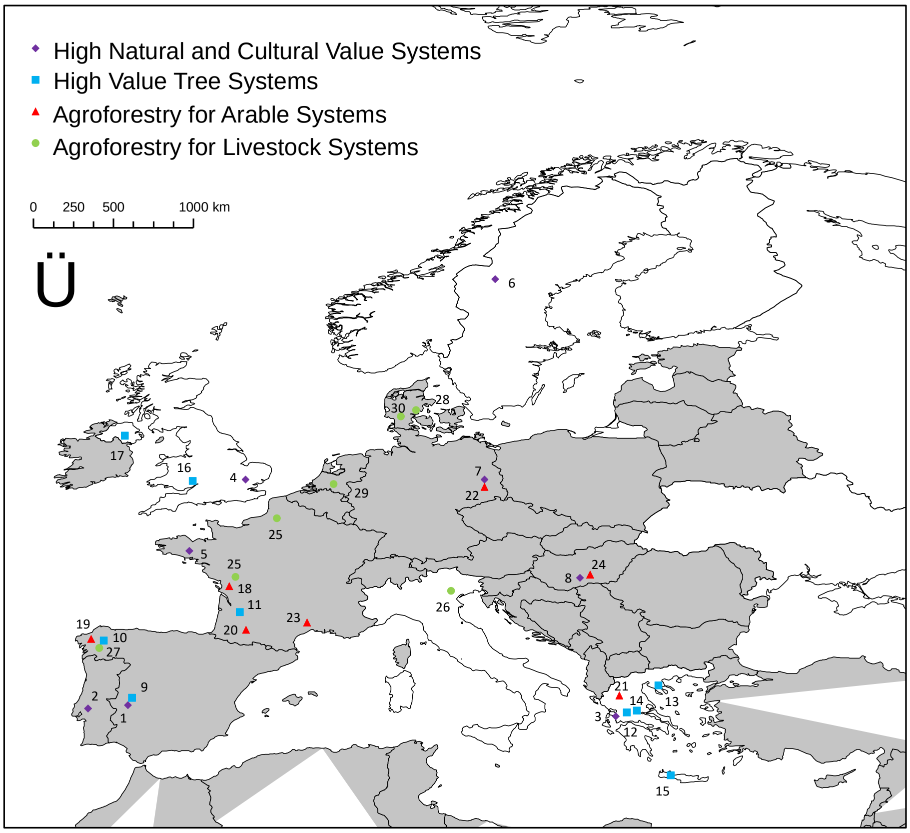

Figure 1. Location of the stakeholder workshops.

Description of the agroforestry systems evaluated

The survey was completed during the initial stage of the AGFORWARD project which seeks to promote appropriate agroforestry in Europe. The systems were grouped according to the aforementioned four agroforestry categories. There were eight surveys completed in the high nature and cultural value agroforestry group, nine in the agroforestry with high value trees group, and seven and six surveys were completed in the agroforestry for arable systems and agroforestry for livestock systems categories respectively. A detailed description of each agroforestry system is provided in **Table 6**.

Table 6. Description of the 30 agroforestry workshops.

System	Description	Number and types of stakeholders and key actors
Agroforestry of high nature and cultural value		
1. *Dehesa*, South-west Spain	Agrosilvo-pastoral systems originating from clearing of evergreen woodlands where trees, native grasses, crops, and livestock interact positively under specific management. The tree species include holm oak (*Quercus ilex* L.) and cork oak (*Quercus suber* L.). Traditional breeds of pigs, cows, sheep and goats are reared at low stocking densities.	67: 26 farmers (livestock breeders), 9 landowners, 16 technical advisors, 5 agrarian administrators, 2 environmentalists, 7 researchers, 2 journalists
2. *Montado,* Portugal	Similar to *dehesa* in Spain but cork oak is usually more abundant	17: 7 technical advisors, 2 farm managers, 2 forest managers, 5 farm and forest managers, 1 other
3. Valonia oak silvopastures in Greece	Silvo-pastoral systems where livestock breeders (sheep and goat) use the valonia oak woodland (*Quercus ithaburensis subsp. macrolepis* (Kotschy)) for grazing. Some acorn cups are used the dye industry.	11: 4 livestock breeders, 2 farmers (livestock breeder) , 1 agronomy student, 4 farmers
4. Wood pasture and parklands in lowland UK	Characterised by veteran trees (often pollarded), grazing livestock, and an understorey of grassland or heathland. Typical tree species include oak, beech and hornbeam.	5: 2 Estate managers; 3 advisors
5. Bocage agroforestry in North-western France	Traditional hedgerow systems largely based on lines of pollarded high-stem trees such as oaks (*Quercus robur* L.), chestnut (*Castanea sativa Mill*.) and beech (*Fagus sylvatica* L.), and medium-stem trees such as hazel (*Corylus avellana* L.) and hornbeam (*Carpinus betulus* L.).	4: 2 farmers, 1 engineer of decentralized State services, 1 technician of a local administration
6. Wood pastures in Northern Sweden	Reindeer husbandry systems based on forest understorey resources. Private forest landowners and enterprises often interact with Sami people, who manage the reindeer, for land-management decisions.	3: 3 Njaarke Sami members (farmers)
7. Agroforestry in Spreewald of Germany	Systems characterized by closely-spaced hedgerows that demarcate individual fields. Common tree species are black alder (*Alnus glutinosa* (L.) Gaertn.), hackberry (*Prunus padus* L.), oak (*Quercus robur* L.) and black poplar (*Populus nigra* L.).	2: 2 farmers
8. Wood pasture, Hungary	Characterised by oak trees (*Quercus robur* L.) with traditional sheep herding.	1: 1 manager of major conservation district of national park
Agroforestry with high value trees		

9. Grazing and intercropping of walnut and cherry, Spain	Plantations of quality timber trees (walnut or cherry) are intercropped with arable crops or grazed by sheep.	27: 10 arable farmers, 7 timber producers, 6 technical advisors, 1 agrarian administrator, 3 academic/researchers
10. Chestnut agroforestry in North-western Spain	Chestnut production is the main focus, but mushrooms and high-quality honey is also harvested. The system is protected by the Natura 2000 network as it is a priority area for birds.	21: 12 chestnut farmers, 2 chestnut processing employees, 5 chestnut association members, 1 expert, 1 rural development member
11. Border trees, South-west France	Managed trees found in rural hedges which often line the side of a road, in riparian forests, buffer strips (with woody vegetation) and wood edges.	10: 3 farmers with border trees, 2 timber producers, 3 riparian technicians, 1 chamber of agriculture, 1 arable farmer
12. Intercropping of walnut trees, Greece	Characterized by walnut trees (*Juglans regia* L.) growing at the edge of fields of maize, dry beans, cereals or pasture.	8: 1 retired farmer, 1 private employee, 6 farmers
13. Intercropping olive groves, Greece	Intercropping of olive (*Olea europaea* L.) groves with arable crops (cereals) to diversify production and income.	13: 1 agronomist, 1 forester, 10 farmers, 1 retired farming employee
14. Grazing and intercropping of olive groves, Greece	Intercropping of olive groves with arable crops (cereals) and grazing with sheep or chicken.	6: 5 farmers, 1 agricultural public servant
15. Intercropping of orange groves, Greece	Intercropping of citrus trees (*Citrus × sinensis* (L.) Osbeck) with intercrops (mainly vegetables) until the tree canopy fully develops, at which stage poultry production can be an option.	5: 3 farmer, 1 agronomist, 1 other
16. Grazed orchards, England, UK	Apple (*Malus domestica* Borkh.) orchards are grazed with sheep. The sheep usually need to be taken out of the orchard during some field operations such as spraying or harvesting. Pears (*Pyrus communis* L.) are also grown.	7: 7 farmers
17. Grazed orchards, N. Ireland, UK	Grazed bramley apple orchards with sheep.	2: 2 apple growers
Agroforestry for arable systems		
18. Silvo-arable agroforestry, Western France	Integration of three to five tree species (e.g. *Juglans regia* L., *Sorbus domestica* L., *Sorbus torminalis* (L.) Crantz, *Prunus avium* L., *Fraxinus excelsior* L., *Acer pseudoplatanus* L., and *Quercus spp*.) in arable fields often with regional government support. Typical tree densities are 30-50 trees per hectare in 27 m rows (24 m cultivated area). Arable crops are often organically managed.	14: 4 farmers and 10 technical advisors
19. Silvo-arable agroforestry, North-western Spain	Widely-spaced trees intercropped with annual or perennial crops.	13: 2 dairy farmers, 2 timber producers, 4 farming cooperative employees, 1 organic producers, 2 representative of rural

		development group, 1 counsellor in farming company, 1 other
20. Silvo-arable agroforestry, South-Western France	Novel methods for integrating trees in crop fields, pastures and vineyards, often with regional government support.	11: 9 agroforestry farmers, 1 member of the chamber of agriculture, 1 local technician for agroforestry plantations
21. Trees with arable crops and grassland, Greece	Trees species such as walnut and poplars grown in the borders of arable fields producing field beans, cereals and grass	10: 3 farmers, 1 forester, 2 agronomists, 2 public servants, 2 farmers
22. Alley cropping, Germany	Experimental system integrating rows of fast-growing trees such as poplar (*Populus spp.*) and black locust (*Robinia pseudoacacia* L.) with arable crops.	6: 1 farmer, 1 retired-farmer, 1 agricultural engineer, 1 landscape architect, 1 researcher, 1 other
23. Silvo-arable agroforestry, Southern France	Integration of trees (e.g. *Populus* species) planted in rows with durum wheat, chickpea, and oilseed rape.	10: 6 farmers, 1 technician, 1 food industry member, 1 organic farmer, 1 seed production advisor
24. Alley cropping in Hungary	Protective shelterbelts, buffer strips and alley cropping on farmsteads or between arable lands	1: 1 managing director of agri-cooperative
	Agroforestry for livestock systems	
25. Agroforestry with ruminants, Northern and mid-Western France	Integration of trees for timber production and as an alternative source of fodder on organic and non-organic grassland and mixed crop-livestock farms with dairy and beef cattle or sheep or goats.	28: 10 farmers, 5 researchers, 10 technical advisors (5 agriculture advisors and 5 agroforestry advisors), 3 others
26. Energy crops and free-range pigs, North-eastern Italy	Free-range pigs with poplar and willow trees for biomass production on paddock borders. The trees provide shade and reduce heat stress during summer months.	22: 9 farmers, 3 members of Dept. of agriculture, 2 veterinarians, 5 agronomist, 3 researchers (forestry and animal science)
27. Pigs with chestnut and oaks, North-western Spain	Semi-extensive or extensive systems focused on pork production in forest areas dominated by chestnut and oak trees.	16: 7 pig breeders, 5 employees in the technological centre of pig, 2 foresters, 1 veterinarian, 1 mushroom mycelia supplier
28. Agroforestry with organic poultry and pigs, Denmark	Organic pig or poultry production on small-holder farms integrated with pasture, fruit trees, bushes and vegetables.	5: 1 organic farmer, 1 private advisor, 1 animal protection member, 1 organic farmer, 1 researcher
29. Fodder trees for cattle and goats, the Netherlands	Fodder trees such as willow are planted for browsing by cattle and goats.	4: 4 farmers
30. Energy crops with free-range pigs, Denmark	Free-range pigs integrated with grass clover crops between rows of short rotation coppice willow (*Salix spp.*) or poplar (*Populus spp.*). Lactating sows are kept outdoors all year round in individual paddocks.	2: 2 organic pig producers

Normalising stakeholders' responses

Each participant was given the same two pages (translated into the local language) which listed issues related to production (9 issues), management (8 issues), the environment (11 issues) and socio-economic issues (17 issues) (Bestman et al. 2014). On the first page, the participants were asked to indicate up to 10 issues that they considered were the most positive aspects of agroforestry (with 1 indicating the highest rank and 2 the second highest rank). On the second page, the participants were asked to indicate the 10 issues which they considered were the most negative. A limitation of this study was that the stakeholder groups used slightly different approaches to rank the positive and negative aspects of agroforestry systems. The groups in Denmark, France, Germany, Italy, the Netherlands, Portugal, Hungary and the UK answered the questionnaire as planned. However at the meetings in Greece (Groups 3, 12, 13, 14, 15, 21), Western Spain (Group 1 and 9) and Sweden (Group 6), most or all of the participants ascribed multiple issues the same ranking, e.g. a participant may have given, for example, ten issues the highest rank value of "1". The three groups in Galicia in North-East Spain (Groups 10, 19 and 27) also used a multiple ranking system, but the ranking was sometimes done within each of the production, management, environment and socio-economic categories, rather than considering the 45 issues as a whole.

The differences in the method of completing the questionnaire meant that it was inappropriate to simply aggregate the stakeholders' responses. To allow comparison between groups, we assumed that where participants only ranked the most positive or negative issues, all of the unranked issues had a low and equivalent rank. For example if the participant only ranked three positive aspects e.g. first rank for biodiversity, second for soil conservation, and third for rural employment then we assumed that participant's ranking scale ranged 1 to 4. We then assumed the ranks for biodiversity, soil conservation and rural employment would be 1, 2, and 3 respectively, and that all of the non-ranked issues were given a value of 4. In this way, all issues were given a rank although the range of ranks could vary with participant. Subsequently, the different ranking ranges were given a normalised rank between 0 and 1 (NR_i) derived from the rank (R_i) given by participant i and the lowest ($Rmin_i$) and highest ($Rmax_i$) rank given that participant (**Equation 1**). Hence in this example, biodiversity and rural employment would have NR values of 0 and 0.67 respectively.

$$\mathrm{NR_i} = \frac{\mathrm{R_i} - \mathrm{Rmax_i}}{\mathrm{Rmin_i} - \mathrm{Rmax_i}} \qquad \text{Equation 1}$$

Finally each normalised rank (NR_i) was subtracted from 1 to create a normalised score (NS_i) so that in the positive issue assessment a higher score indicates a more positive issue and in the negative issue assessment, higher values indicated higher negative values.

$$\mathrm{NS_i} = 1 - \mathrm{NR_i} \qquad \text{Equation 2}$$

Results

This study describes how stakeholders scored the negative and positive aspects of implementing agroforestry practices. The results are presented first in terms of the overall mean result, and then in terms of four categories of agroforestry systems and the 30 individual groups.

Overall results

The results were first analysed in terms of the overall effect and the same weight was given to each system e.g. the response from the *dehesa* in Spain (67 respondents) is given the same weight as wood pasture in Hungary (1 respondent). A higher mean normalised positive score was achieved for environmental (0.31) and production (0.31) issues than management (0.20) and socio-economic (0.16) issues (**Figure 2**). In terms of specific issues, the highest normalised positive scores were achieved for biodiversity and wildlife habitat (0.53), animal health and welfare (0.48), landscape aesthetics (0.43), general environment (0.39), soil conservation (0.39) and diversity of products (0.37).

In terms of negative issues, the highest mean normalised score was obtained for management issues (0.23), followed by socio-economic (0.12) and production (0.10), with environmental issues (0.06) of lowest concern. The highest individual normalised negative scores were achieved for labour (0.35), administrative burden (0.32), complexity of work (0.31) and management costs (0.31).

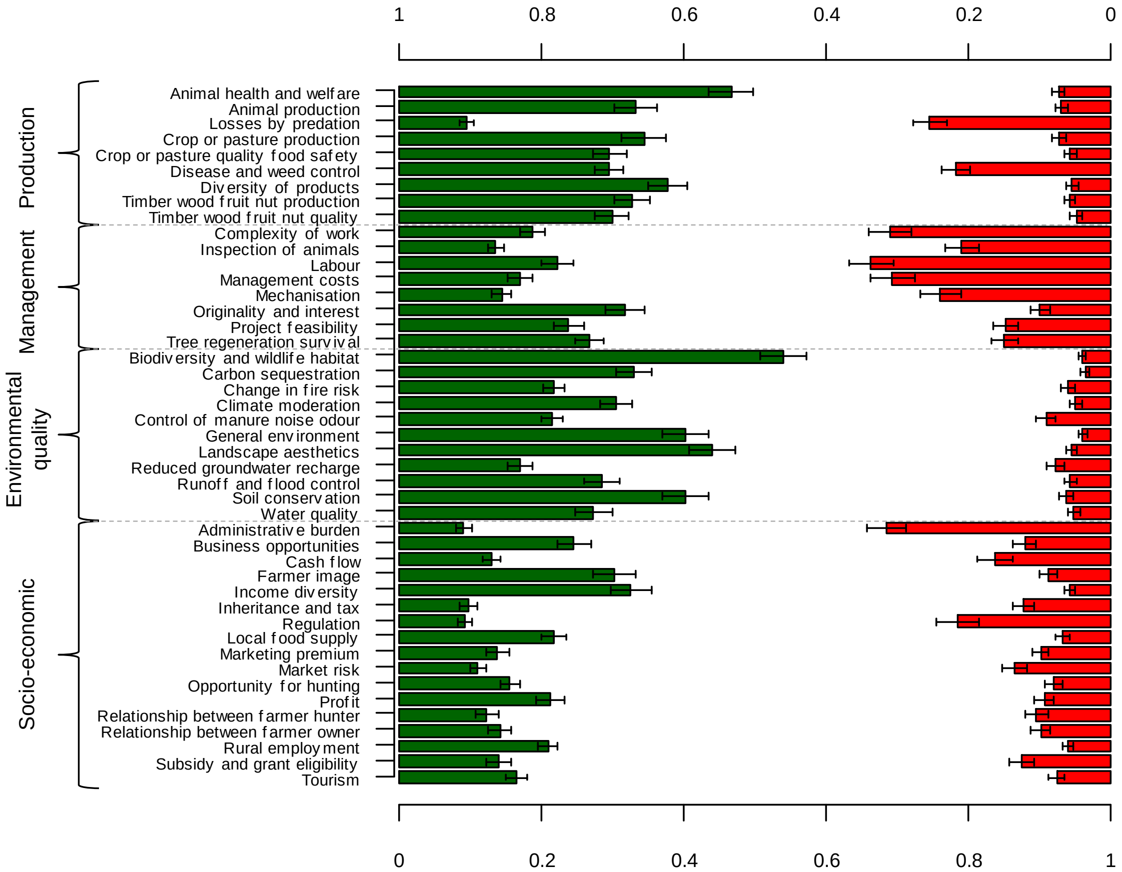

Figure 2. Mean normalised scores received from 30 stakeholder groups (comprising 344 stakeholders) on the positive (green bars on the left) and negative issues (red bares on the right) related to selected agroforestry systems across Europe. Error bars indicate the 95% confidence interval for the estimated mean.

Results per agroforestry category

The mean normalised score received for each issue within each of the four categories of agroforestry system are described in **Table 7**. The three individual positive and negative issues receiving the highest normalised score in each of the 30 groups are presented in **Tables 8** and **Table 9**.

Table 7. Positive and negative aspects of the categories of agroforestry in Europe.

Criteria	Aspects	Positive assessment				Negative assessment			
		High Natural and Cultural Value systems	High Value Tree systems	Agroforestry for Arable systems	Agroforestry for Livestock systems	High Natural and Cultural Value systems	High Value Tree systems	Agroforestry for Arable systems	Agroforestry for Livestock systems
Production	Animal health and welfare	0.35	0.51	0.32	0.71	0.07	0.07	0.04	0.13
	Animal production	0.35	0.41	0.26	0.26	0.02	0.13	0.05	0.09
	Losses by predation	0.05	0.16	0.10	0.05	0.34	0.23	0.23	0.20
	Crop or pasture production	0.33	0.40	0.47	0.14	0.01	0.10	0.11	0.07
	Crop or pasture quality food safety	0.22	0.45	0.28	0.18	0.03	0.09	0.05	0.06
	Disease and weed control	0.30	0.45	0.19	0.17	0.24	0.10	0.25	0.31
	Diversity of products	0.37	0.51	0.24	0.34	0.04	0.07	0.07	0.04
	Timber wood fruit nut production	0.18	0.51	0.33	0.23	0.01	0.10	0.08	0.03
	Timber wood fruit nut quality	0.26	0.46	0.23	0.19	0.02	0.08	0.04	0.06
Management	Complexity of work	0.22	0.27	0.10	0.11	0.19	0.43	0.30	0.33
	Inspection of animals	0.14	0.21	0.07	0.09	0.23	0.27	0.10	0.24
	Labour	0.25	0.34	0.13	0.11	0.26	0.24	0.41	0.49
	Management costs	0.11	0.33	0.09	0.10	0.33	0.30	0.32	0.27
	Mechanisation	0.12	0.23	0.09	0.10	0.13	0.23	0.34	0.28
	Originality and interest	0.31	0.40	0.21	0.32	0.01	0.17	0.10	0.12
	Project feasibility	0.25	0.32	0.21	0.13	0.07	0.14	0.30	0.09
	Tree regeneration survival	0.30	0.41	0.16	0.14	0.10	0.14	0.16	0.23
Environmental quality	Biodiversity and wildlife habitat	0.61	0.60	0.41	0.50	0.01	0.06	0.04	0.06
	Carbon sequestration	0.31	0.50	0.26	0.19	0.00	0.07	0.02	0.05
	Change in fire risk	0.20	0.38	0.11	0.12	0.02	0.10	0.08	0.04
	Climate moderation	0.25	0.44	0.38	0.09	0.01	0.08	0.05	0.08
	Control of manure noise odour	0.15	0.44	0.09	0.10	0.08	0.08	0.10	0.11
	General environment	0.37	0.53	0.37	0.31	0.01	0.06	0.05	0.04
	Landscape aesthetics	0.45	0.58	0.25	0.44	0.00	0.08	0.10	0.05
	Reduced groundwater recharge	0.11	0.34	0.12	0.06	0.07	0.10	0.09	0.05
	Runoff and flood control	0.26	0.44	0.19	0.19	0.03	0.07	0.07	0.06
	Soil conservation	0.32	0.55	0.50	0.17	0.04	0.07	0.08	0.06
	Water quality	0.21	0.40	0.29	0.14	0.01	0.07	0.04	0.10
Socio-economic	Administrative burden	0.10	0.16	0.03	0.04	0.31	0.31	0.27	0.39
	Business opportunities	0.26	0.32	0.19	0.18	0.04	0.21	0.14	0.08
	Cash flow	0.15	0.20	0.08	0.06	0.15	0.18	0.22	0.08
	Farmer image	0.29	0.39	0.21	0.29	0.03	0.18	0.10	0.04
	Income diversity	0.25	0.43	0.41	0.17	0.01	0.11	0.06	0.05
	Inheritance and tax	0.14	0.11	0.06	0.05	0.06	0.17	0.17	0.09
	Regulation	0.12	0.09	0.08	0.05	0.15	0.27	0.20	0.24
	Local food supply	0.16	0.35	0.15	0.17	0.08	0.07	0.06	0.05
	Marketing premium	0.17	0.16	0.07	0.13	0.02	0.16	0.14	0.06
	Market risk	0.14	0.17	0.06	0.05	0.04	0.18	0.18	0.15
	Opportunity for hunting	0.19	0.22	0.13	0.06	0.02	0.16	0.10	0.03
	Profit	0.18	0.37	0.15	0.09	0.10	0.12	0.08	0.07
	Relationship between farmer hunter	0.13	0.16	0.10	0.08	0.05	0.18	0.10	0.08
	Relationship between farmer owner	0.14	0.17	0.12	0.12	0.05	0.15	0.11	0.08
	Rural employment	0.20	0.34	0.13	0.11	0.02	0.13	0.05	0.04
	Subsidy and grant eligibility	0.14	0.17	0.16	0.06	0.07	0.13	0.11	0.20
	Tourism	0.21	0.21	0.08	0.13	0.02	0.15	0.09	0.04

Agroforestry of high nature and cultural value

In agroforestry systems of high nature and cultural value, the highest positive normalised score was received for enhanced biodiversity and wildlife habitat (0.61) (This was the highest-ranking issue in the hedgerow agroforestry systems in France and Germany and the wood pasture system in the UK (**Table 8**). The next highest score was for landscape aesthetics (0.45) and this was the highest-ranking issue in the *dehesa* system in Spain. The broad term "general environment" received a score of 0.37, followed by diversity of products (0.37), animal health and welfare (0.35) and animal production (0.35). Animal health and welfare was ranked the highest positive issue in the silvo-pastoral systems in Greece. Although not ranked highest across the eight systems as a whole, income diversity was the most important positive aspect in Portugal, rural employment was ranked highest in the reindeer silvo-pastoral system in Sweden, and disease and weed control was identified as the most positive aspect of wood pasture in Hungary.

Table 8. Three issues receiving the highest normalised positive score in each of 30 studied agroforestry systems.

	Systems	Highest score	Second	Third	n
High nature and cultural value	1. *Dehesa*, South-west Spain	Landscape aesthetics	General environment	Soil conservation	67
	2. *Montado*, Portugal	Income diversity	Biodiversity and wildlife habitat	Diversity of products	17
	3. Valonia oak silvopastures, Greece	Animal health and welfare	Animal production	Diversity of products	11
	4. Wood pasture and parklands, UK	Biodiversity and wildlife habitat	Soil conservation	Landscape aesthetics	5
	5. Bocage agroforestry, France	Biodiversity and wildlife habitat	Carbon sequestration	Runoff and flood control	4
	6. Wood pastures in Northern Sweden	Rural employment	Business opportunities	General environment	3
	7. Agroforestry in Eastern Germany	Biodiversity and wildlife habitat	Crop or pasture production	Diversity of products	2
	8. Wood pasture in Hungary	Disease and weed control	Biodiversity and wildlife habitat	Runoff and flood control	1
Agroforestry with high value trees	9. Grazing and intercropping of walnut and cherry, Spain	General environment	Landscape aesthetics	Soil conservation	27
	10. Chestnut agroforestry, North-west Spain	Biodiversity and wildlife habitat	Diversity of products	Tree regeneration survival	21
	11. Border trees, South-western France	Timber, wood, fruit and nut production	Biodiversity and wildlife habitat	Landscape aesthetics	10

	12. Intercropping of walnut trees, Greece	Diversity of products	General environment	Landscape aesthetics	8
	13. Intercropping of olive groves, Greece	Timber, wood, fruit and nut quality	Biodiversity and wildlife habitat	Diversity of products	13
	14. Grazing and intercropping of olive groves, Greece	Animal health and welfare	Control of manure, noise and odour	Timber, wood, fruit and nut production	6
	15. Intercropping of orange groves, Greece	Runoff and flood control	Soil conservation	Crop or pasture quality food safety	5
	16. Grazed orchards, England, UK	Animal production	Labour	Management costs	7
	17. Grazed orchards, Northern Ireland, UK	Animal health and welfare	Profit	Crop or pasture production	2
Agroforestry for arable systems	18. Silvo-arable agroforestry, Western France	General environment	Biodiversity and wildlife habitat	Soil conservation	14
	19. Silvo-arable agroforestry, North-western Spain	Business opportunities	Originality and interest	Project feasibility	13
	20. Silvo-arable agroforestry, South-Western France	Timber, wood, fruit and nut production	Soil conservation	Biodiversity and wildlife habitat	11
	21. Trees with arable crops and grassland, Greece	Animal health and welfare	Timber, wood, fruit and nut quality	Animal production	10
	22. Alley cropping, Germany	Crop or pasture production	Soil conservation	Landscape aesthetics	6
	23. Silvo-arable agroforestry, Southern France	Income diversity	Crop or pasture production	Biodiversity and wildlife habitat	3
	24. Alley cropping, Hungary	Climate moderation	Crop or pasture production	Income diversity	1
Agroforestry for livestock	25. Agroforestry with ruminants, Northern and mid-Western France	Animal health and welfare	Farmer image	Biodiversity and wildlife habitat	28
	26. Energy crops and free-range pigs, North-eastern Italy	Diversity of products	Animal health and welfare	Timber, wood, fruit and nut quality	22
	27. Pigs with chestnuts and oaks, North-western Spain	Biodiversity and wildlife habitat	Project feasibility	Tree regeneration survival	16
	28. Agroforestry with organic poultry and pigs, Denmark	Animal health and welfare	Diversity of products	Biodiversity and wildlife habitat	5
	29. Fodder trees for cattle and goats, the Netherlands	Animal health and welfare	Landscape aesthetics	Biodiversity and wildlife habitat	4
	30. Energy crops with free-range pigs, Denmark	Animal health and welfare	Biodiversity and wildlife habitat	Landscape aesthetics	2

In terms of negative aspects, agroforestry of high nature and cultural value was seen to result in losses due to predation (0.34) and this was the dominant negative issue in Greece and Hungary (**Table 9**). Management costs (0.33) and labour (0.26) were the main negative effects in terms of management, with labour being the highest ranked negative issue by the French and German group, and management costs ranked second in France, Germany, and Sweden. Administrative burden (0.31) was seen as the main negative socio-economic issue and it received the highest negative ranking in Spain. Other issues that were ranked highest by individual groups were complexity of work in the UK and regulation in Portugal.

Table 9. Three issues receiving the highest normalised negative score in each of 30 studied agroforestry systems.

	Systems	Highest score	Second	Third	n
High nature and cultural value	1. *Dehesa*, South-west Spain	Administrative burden	Subsidy and grant eligibility	Mechanisation	66
	2. *Montado*, Portugal	Regulation	Tree regeneration survival	Complexity of work	15
	3. Valonia oak silvopastures, Greece	Losses by predation	Reduced groundwater recharge	Soil conservation	7
	4. Wood pasture and parklands, UK	Complexity of work	Inspection of animals	Management costs	5
	5. Bocage agroforestry, North-western France	Labour	Management costs	Cash flow	4
	6. Wood pastures in Northern Sweden	Disease and weed control	Management costs	Losses by predation	3
	7. Agroforestry in Germany	Labour	Management costs	Administrative burden	2
	8. Wood pasture in Hungary	Losses by predation	Administrative burden	Inspection of animals	1
Agroforestry with high value trees	9. Grazing and intercropping of walnut and cherry, Western Spain	Administrative burden	Subsidy and grant eligibility	Mechanization	27
	10. Chestnut agroforestry, North-western Spain	Complexity of work	Animal production	Losses by predation	21
	11. Border trees, South-west France	Lack of knowledge	Management costs	Mechanisation	10
	12. Intercropping of walnut trees, Greece	Marketing premium	Cash flow	Business opportunities	8
	13. Intercropping olive groves, Greece	Administrative burden	Management costs	Complexity of work	10
	14. Grazing and intercropping of olive groves in Greece	Losses by predation	Opportunity for hunting	Relationship between farmer hunter	7
	15. Intercropping of orange groves, Greece	NA	NA	NA	5
	16. Grazed orchards, England, UK	Complexity of work	Inspection of animals	Management costs	7
	17. Grazed orchards, N. Ireland, UK	Complexity of work	Cost of fencing boundary	Inspection of animals	2
Agroforest ry for	18. Silvo-arable agroforestry, Western France	Complexity of work	Labour	Cash flow	14
	19. Silvo-arable agroforestry, North-western Spain	Complexity of work	Losses by predation	Mechanisation	13

	20. Silvo-arable agroforestry, South-Western France	Management costs	Project feasibility	Administrative burden	10
	21. Trees with arable crops and grassland, Greece	Management costs	Losses by predation	Labour	10
	22. Alley cropping, Germany	Labour	Business opportunities	Cash flow	3
	23. Silvo-arable agroforestry, Southern France	Regulation	Administrative burden	Management costs	3
	24. Alley cropping, Hungary	Disease and weed control	Project feasibility	Labour	1
Agroforestry for livestock	25. Agroforestry with ruminants, Northern and mid-Western France	Complexity of work	Labour	Mechanisation	28
	26. Energy crops and free-range pigs, North-eastern Italy	Tree regeneration survival	Inspection of animals	Complexity of work	22
	27. Pigs with chestnuts and oaks, North-western Spain	Administrative burden	Losses by predation	Animal production	12
	28. Agroforestry with organic poultry and pigs, Denmark	Labour	Complexity of work	Administrative burden	5
	29. Fodder trees for cattle and goats, the Netherlands	Disease and weed control	Labour	Tree regeneration survival	3
	30. Energy crops with free-range pigs, Denmark	Labour	Administrative burden	Management costs	2

Agroforestry with high value trees

For agroforestry related to high value trees, the mean normalised scores for positive issues tended to be greater than for the other three categories of systems. This is a result of the majority of these groups (primarily in Greece and Spain) allowing multiple first and second rankings. The highest positive values were again received for the enhancement of biodiversity and wildlife (0.60) and improved landscape aesthetics (0.58). Enhancement of biodiversity was ranked highest in Spain and ranked second in France and by one of the Greek groups. Soil conservation (0.55), the general environment (0.53), and carbon sequestration (0.50) was also ranked high across the eight groups. Reducing runoff and flood control was ranked the most positive aspect by the orange intercropping group in Crete, Greece. High scores were also received for various aspects of production including the production of timber wood, fruit and nuts (0.51), diversity of products (0.51), and animal health and welfare (0.51). Production of tree products was the most important positive issues for one group in Greece and the group in France. Product diversity was ranked highest by the walnut intercropping group in Greece where the products included walnuts, timber, maize, vegetables, and beans. Animal welfare was considered the most positive issue with another group in Greece and the grazed orchard group in Northern Ireland in the UK. The other issue ranked highest by an individual group was animal production by the grazed orchard group in England, UK. The positive scores received for the individual management and socio-economic issues were less than 0.44.

In terms of negative issues, the most important aspect was the complexity of work (0.43). This was also individually identified as the greatest negative issue in North West Spain, and the two grazed orchard systems in the UK. The next most significant issues were the administrative burden (0.31) and management costs (0.30). The administrative burden was ranked as the most important negative issue in one Greek and one Spanish site. Management costs were considered to be the second most important negative issue by the French and one of the Greek groups. At an individual group level, a lack of knowledge was considered the most important negative issues by the French group dealing with border trees, and losses by predation was ranked highest by one of the olive agroforestry groups in Greece. The lack of a marketing premium was also highlighted by the walnut intercropping group in Greece.

Agroforestry for arable systems

In terms of agroforestry for arable systems, each of the seven individual groups identified a different issue as the most important benefit of agroforestry. This suggests that the key advantage of agroforestry within an arable system is less clear than with the other categories. The highest positive normalised score was for soil conservation (0.50). Although no individual group identified this as the most important feature; it was ranked second or third in Southwest France, Western France, and Germany. The second highest score was achieved for crop production (0.47) and this was the most highly ranked issue with the German group and was ranked second by the groups in Southern France and Hungary. The third highest scores were for income diversity (0.41) and an enhanced biodiversity and wildlife habitats (0.41). Income diversity was ranked highest in southern France, and biodiversity benefits were ranked in the top three in south-west and western France. Other issues that were ranked highest by an individual group were timber, wood, fruit and nut production in South-West France and business opportunities in Northwest Spain. Climate moderation was ranked as the highest positive issue in Hungary where the focus was on the use of trees for shelterbelts. The highest ranked issue for the Greek group was improved animal health and welfare, which suggests that although the Greek group was included under "arable systems", the wide extent of mixed farms meant that animal welfare remains important on farms producing arable crops in Greece.

The five highest ranked negative issues all relate to management, namely labour (0.41), mechanisation (0.34), management costs (0.32), complexity of work (0.30) and project feasibility (0.30). Labour was ranked as the greatest constraint by the silvo-arable group in southern France and was ranked in the top three by the groups in Western France, Greece and Hungary. Mechanisation was ranked third in North-West Spain, and management cost was the most critical issue in Western France and Greece. Complexity of work was the major issue in Western France and North-West Spain. The other two negative issues that scored highest within an individual group was regulation in Germany and disease and weed control in Hungary.

Agroforestry for livestock systems

There were six groups focused on agroforestry for livestock and these groups generally gave similar responses. The highest positive score for an issue, and in fact the highest score for any issue across the four agroforestry categories, was for animal health and welfare (0.71). This was also the highest positive factor in four of the six groups i.e. two groups in Denmark and the groups in France and the Netherlands, and it was ranked second with the group from Italy. The second highest positive score was in terms of enhanced biodiversity and wildlife habitats (0.50) and this was identified as the most important issue in North-West Spain. Across the six groups the third highest score (0.44) was for improved landscape aesthetics, which had a top three ranking from the group in the Netherlands and the free-range pig group in Denmark. The energy crops for free-range pigs' group in Italy identified the diversity of products as the most important issue.

Increased labour (0.49) was seen as the most negative issue, and in fact this received the highest negative score for an individual issue within an agroforestry category. It was also the highest ranked constraint by the two groups in Denmark and was ranked second in Western France and the Netherlands. This was also associated with increased administrative burden (0.39), which was ranked first by the group in North West Spain and second by the free-range pig group in Denmark. Across the category the third ranking was given to the complexity of work (0.33), and this was seen as a top three issue in Western France, Italy, and a group in Denmark. The fourth most important issue was disease and weed control (0.33), and this was particularly highlighted by the group in the Netherlands in relation to tree establishment. The group in Italy considered that tree survival was a major issue, and this was also identified by the group in the Netherlands working with goats.

Discussion

Motivations to undertake agroforestry

The study has highlighted four key drivers motivating the practice of agroforestry: biodiversity, soil conservation, enhanced animal health and welfare, and income diversity. These are discussed in turn.

Biodiversity and landscape aesthetics: in the agroforestry with high nature and cultural value and agroforestry with high value trees categories the enhancement of biodiversity and wildlife habitats was the dominant positive attribute. Most of the high nature and cultural value agroforestry systems were wood pastures which are widely recognised in Europe for their high ecological value (Plieninger et al. 2015). Campos Palacín and Mariscal Lorente (2003) showed that *dehesa* owners often value more self-consumption of recreational and environmental services such as landscape aesthetics and biodiversity than marketed farm products. Some of the systems considered as agroforestry for high value trees, such as the chestnut system in North West Spain, are also valued in terms of their biodiversity and are protected Natura 2000 sites. The high scores related to landscape aesthetics also highlight that these agroforestry systems are not just valued in terms of their ecology, but also their cultural importance. There is evidence that people prefer to see diversified landscapes with trees than without trees (Kaplan and Talbot 1988; Gómez-Limón and Lucío Fernández 1999; Herzog et al. 2000).

Soil conservation: in agroforestry for arable systems, the key positive motivation was the combination of maintaining crop production with soil conservation. Particularly in silvo-arable alley cropping systems soil conservation was seen as a key environmental benefit. Soil loss is a major factor determining the long-term productivity of many arable farms. For example a recent study in the UK has highlighted that soil degradation could have an annual cost of £1.2 billion with about half related to the loss of soil organic matter, 40% to compaction, and 12% to soil erosion (Graves et al. 2015). In terms of supporting agroforestry, a focus on soil conservation may be particularly useful in that the benefits can be tangible at the farm level (e.g. improved productivity and reduced soil management costs) and, in addition, provide benefits at a wider landscape scale (e.g. reduced flooding and water purification costs).

Animal health and welfare: in agroforestry systems focused on livestock production (e.g., energy crops with free-range pigs and agroforestry with organic poultry), the key motivation was improved animal health and welfare. Broom et al. (2013) has highlighted the positive effect of trees on animal welfare by providing shade from hot sun and shelter from precipitation and extreme cold temperatures. Hens, which are a species adapted to tree cover, can also show more natural behaviour when given access to trees.

Diversity of products and income diversity: diversifying sources of farm income is a key motivation for more risk-averse farmers. Similar to our results, Graves et al. (2009) also found that stakeholders perceived diversity of products to be a major benefit of silvo-arable systems.

Constraints to undertake agroforestry

The analysis demonstrates that the key constraints to implementing agroforestry often relate to management issues. In broad terms the same constraints occurred across the four categories of agroforestry namely: high labour requirements, complexity of work, management costs and administrative burdens. Loss by predation was also highlighted within the agroforestry for high nature and cultural value category.

Labour: A key driver in agricultural decisions is the need to increase labour productivity. For example between 1953 and 2000, whilst output per unit area in the UK doubled, the output per unit labour increased at least five-fold (Burgess and Morris 2009). In some situations, this increase in labour productivity resulted in higher wages, but there can sometimes be a cost to social interaction and the number of people employed on farms.

In silvo-arable alley cropping systems and agroforestry systems focused on livestock production a key barrier to adoption was the increased labour requirements. Compared to livestock production with no tree cover, agroforestry can require more labour due to tree management operations and difficulties in machinery use (Brownlow et al. 2005). On the other hand, higher labour requirements can lead to an increase in jobs in rural areas which is an important goal of EU policies.

Complexity of work and management costs: these were perceived as important barriers to the implementation of agroforestry in Europe. The management of agroforestry systems can be more complex than conventional agriculture as managers need to consider a wider range of variables, for example the management of the tree component and the phasing of crop, livestock and tree operations (Pannell 1999).

Increased complexity can be an important aspect to consider when livestock are incorporated into high value tree systems such as fruit orchards and olive groves. For example, whilst the introduction of sheep to an apple orchard can increase overall revenue, the integrated management requires the manager to have both tree and livestock management skills or for the orchard manger to work with a sheep farmer. The orchard manager and sheep farmer also need to address management constraints such as the need to remove sheep from the orchard for approximately 60 days before apple harvest to prevent faecal contamination.

Administrative burden: several stakeholders identified that the Common Agricultural Policy (CAP) of the EU disadvantaged agroforestry relative to conventional agricultural systems. (Eichhorn et al. 2006) also identified that the CAP played a major role in the recent decline of silvo-arable agroforestry systems across Europe. The high administrative burden associated with agroforestry could be a result of the CAP itself or individual national interpretations of the CAP. For example, stakeholders in Spain highlighted that the management of the *dehesa* wood pasture system required higher levels of administrative input than conventional arable agriculture. Furthermore, they claimed difficulties for getting permission for pruning, an excess of permission for transhumance and lack of efficient green accounting systems for multipurpose systems.

Methods to promote agroforestry

Producers and landowners considering agroforestry need to believe that the benefits outweigh the extra costs involved in the implementation and maintenance of agroforestry systems. Four key methods for promoting agroforestry include i) national demonstration sites, ii) improved regulation, iii) providing a market for the positive externalities with agroforestry, and iv) increasing the opportunities for new profitable businesses.

National demonstrations and education: education, training programmes and use of demonstration sites could play a key role in overcoming the barriers associated with operational complexity. Following the requirements for adoption as identified by Pannell (1999), farmers first need to be able to select the most appropriate agroforestry practice, perceive that the practice is feasible to trial, perceive that the innovation is worth trialing, and feel that the practice promotes their objectives. The use of demonstration sites and field days organized by extension services could be used to introduce farmers to novel agroforestry practices and compare and show their advantages over other systems.

Improved regulation: some of the administrative burden associated with agroforestry can be addressed through simplified and/or improved policies. At present it is argued that there are complex regulations that lead to simplified landscapes; is it possible to have simplified regulations that lead to more diversed landscapes? For example in the *dehesa*, farmers highlighted the difficulty of retaining full eligibility of wood pastures for Pillar I CAP payments. One potential way forward is for managers of agroforestry systems to work with national farming associations to improve communication with policy makers at local, national and EU level.

Market for positive externalities: many of the benefits of agroforestry are environmental which are non-market benefits, and hence agroforestry farmers are not financially compensated for the societal benefits that they provide. Moreover, some of these "non-market benefits" occur not just on-farm but at a wider landscape or catchment scale. Since currently, it is often only market costs and benefits that are guiding decision-making it is argued that this has led to sub-optimal land uses from a societal perspective, and hence (with due care) there may be a case for government and, for example, utility companies to compensate farmers who integrate trees with farming. In some cases, awareness alone of the environmental benefits is insufficient to lead to the adoption of conservation practices (Knowler and Bradshaw 2007). Farmers need to perceive that the practice will provide benefits on their own farm or that they will be compensated for the extra costs (Greiner and Gregg 2011). To some extent, the magnitude of the environmental benefit perceived by each person depends on personal knowledge, awareness and attitudes towards the environment (Jacobsen et al. 2008). A farmer with low environmental awareness is therefore less likely to adopt agroforestry practices than a farmer with high environmental awareness (Reimer et al. 2012; García de Jalón et al. 2013). Thus, raising farmers' environmental awareness could be an additional approach to promoting agroforestry practices.

Profitable business opportunities: many agricultural innovations are founded on the business opportunity of improved profit. In this study, the business opportunities and the profit associated with agroforestry were not seen as key drivers. Workman et al. (2003) highlighted lack of markets as a barrier to the adoption of agroforestry. One of the key areas where agroforestry systems have recently been adopted in the UK is in relation to woodland eggs and chickens driven by an increase in societal concern about farm animal welfare (Jones et al. 2007). In this case, consumers and NGOs have perceived that a welfare benefit for hens and other poultry exists when they have access to a wooded environment, and hence specific labels or contracts may specify that that poultry owners need to provide access for their stock to woodland.

Conclusions

The main positive aspects of agroforestry as perceived by stakeholders in Europe were primarily environmental or production-based, with specific benefits being enhanced biodiversity and wildlife habitats, landscape aesthetics, soil conservation, and animal health and welfare. By contrast, the main negative aspects of agroforestry were primarily related to management and socio-economic issues, with the principal constraints being increased labour, complexity of work, management costs, the administrative burden and in some cases predation by wild animals.

Successful adoption and maintenance of agroforestry systems requires farmers to perceive that the net benefit provided by agroforestry is greater than alternative land use options. If there is clear quantification of the environmental benefits provided by agroforestry, then there is a case for national governments, NGOs and motivated individuals to use education, regulation, market mechanisms and marketing innovation to promote wider adoption and maintenance of agroforestry systems.

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Chapter 3| Modelling the tree density effects on provisioning ecosystem services in Europe

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Abstract

Agroforestry systems, in which trees are integrated in arable or pasture land, can be used to enable sustainable food, material, and energy production (i.e. provide provisioning ecosystem services) whilst reducing the negative environmental impacts associated with farming. However, one constraint on the uptake of agroforestry in Europe is a lack of knowledge on how specific agroforestry designs affect productivity. A process-based bio-physical model, called Yield-SAFE, was used: 1) to quantify the food, material and biomass energy production of four contrasting case study systems in Europe in a common energy unit (MJ ha^{-1}), and 2) to quantify how tree density determined the supply of provisioning ecosystem services. The Yield-SAFE model was calibrated so that simulated tree and crop growth fitted observed growth data for reference monoculture forestry, pasture, and arable systems. The modelled results showed that including trees in pasture or arable systems increased the overall accumulated energy of the system in comparison with monoculture forestry, pasture, and arable systems, but that the accumulated energy per tree was reduced as tree density increased. The greatest accumulated energy occurred in the highest tree density agroforestry system at all the case study sites. This suggests that the capture of environmental resources, such as light and water, for obtaining provisioning services is most effective in high density agroforestry systems. Further modelling should include tree canopy effects on micro-climatic and the impact this has on pasture, crop, and livestock yields, as well as the impact of tree density on the economic value and management of the different systems.

Keywords

Yield-SAFE, agroforestry systems, *montado, dehesa*, short rotation coppice, silvo-arable, silvopasture.

Introduction

Global population growth, combined with rising levels of consumption, are increasing the demand for natural resources and pressure on the environment. In this context, agroforestry systems, where trees are integrated in arable or pasture land, have received considerable attention in both tropical (Garrity et al. 2010) and temperate regions (Palma et al. 2007b). They are increasingly seen as a promising approach for improving food, energy, and material provision (Glover et al. 2012) along with environmental conservation and stimulation of local economies (IAASTD 2009). The components of agroforestry systems can be complementary in their use of solar radiation and water, leading to an overall higher biomass production than when the same components are grown in separate tree, pasture, or arable systems (Graves et al. 2007). Rivest et al. (2013) also suggested that tree presence has a major role to play in landscapes, acting as a keystone structure for maintaining ecosystem services. However, Torralba et al. (2016) reported in a literature meta-analysis that no clear effect of agroforestry on provisioning services could be determined, partly because of the different ways in which provisioning services can be defined e.g. crop production or combined crop and tree production. Smith et al. (2013) reported a reduction of arable crop yields in agroforestry systems when physical resources such as light in temperate regions (Chirko et al. 1996; Reynolds et al. 2007; Benavides et al. 2009) or water in the Mediterranean basin (Jose et al. 2004) were limiting. However, the advantages and disadvantages of agroforestry are dependent on site-specific responses by trees, crops, and other components of the system, with large variation between locations and farming contexts (Coe et al. 2014). There is a complex relationship between climate, soil water content, water uptake by trees and crops, plant growth and evaporation, which challenges our understanding of water, radiation and growth dynamics in agroforestry systems. This uncertainty relating to the potential productivity of agroforestry has been suggested as one of the main causes for the low implementation of new agroforestry systems (Pisanelli et al. 2014).

The use of models can help develop knowledge on how different components of an agroforestry system interact in space and time. Furthermore, process-based models can be used to analyse management decisions in uncertain climatic conditions (Cuddington et al. 2013). The Yield-SAFE model is a process-based growth model that evaluates competition between trees and crops for water and light (van der Werf et al. 2007) which has been extensively used to predict tree and crop yields in arable, agroforestry and forestry systems. Within the AGFORWARD project (Burgess and Rosati 2018) Yield-SAFE was calibrated for several new agroforestry systems in Europe and improved with new algorithms to predict the output of additional provisioning ecosystem services such as pasture, root crops and tree fruit production (Palma et al. 2016).

The objectives of this study were: 1) to quantify in energy units (e.g. MJ ha^{-1}) the food, material and biomass energy production of four contrasting case study agroforestry systems in Europe, and 2) to improve the understanding of how tree density determines the supply of provisioning ecosystem services. The Yield-SAFE model was used on four sites in different parts of Europe and for each site, six different land use alternatives of differing tree densities were considered: a crop rotation or pasture (zero tree density); four agroforestry systems (intermediate tree densities) and a tree-only system (high tree density).

Materials & Methods

Several steps are needed for the use of the Yield-SAFE model. These include: 1) the identification and description of the crop, pasture, agroforestry and tree-only systems to be analysed; 2) the collection of site data on weather, soil texture and depth, and management practices for the crops, pasture, livestock and trees; 3) the identification, often from literature, of parameter values for the crops, pasture, livestock and trees for the calibration of the model, and; 4) calibration of the model outputs against field data. The full details of this process are provided in Graves et al., (2007).

For this study, the simulation period was 80 years for all the case study systems. The provisioning ecosystem services were classified according to their origin (tree, crop and livestock) and their use (food, energy or materials) and converted into energy units (MJ ha^{-1}) using the Utilisable Metabolizable Energy value (UME in MJ kg^{-1}) for food products and the gross calorific value (GCV in MJ kg^{-1}) for energy and materials. The use of an energy unit (MJ ha^{-1}) allowed for standardization of provisioning ecosystem services outputs, thereby allowing comparison of the total food, material, and energy produced by the different systems during the simulation period, so that the effects of the different tree densities could be assessed.

Identification and description of the case study systems and the provisioning ecosystem services supplied

Four different types of agroforestry systems were selected to represent different environmental conditions across Europe. These systems were: 1) Iberian wood pastures (*dehesa* in Spain and *montado* in Portugal); 2) cherry tree pastures in Switzerland (Swiss orchards); 3) poplar silvo-arable systems in the United Kingdom and; 4) short rotation poplar coppice systems for biomass energy in Germany. The meteorology, soil conditions and the agroforestry components for each system are described in **Table 10**.

Table 10. Location, meteorological and soil information and component description of the four agroforestry systems studied.

	Montado	**Cherry tree pastures**	**Silvo-arable systems**	**SRC**
Location	Montemor-o-Novo,PT	Gempen, CH	Silsoe, UK	Forst, DE
Identification	MONTPT	CTCH	SAFUK	SRCDE
Altitude (m asl)	130	680	70	75
Longitude (°)	38.7023	7.2299	50.0089	51.7890
Latitude (°)	-8.3261	6.9943	0.4358	14.4918
Meteorological conditions				
Mean annual solar radiation (MJ m^{-2})	6080	4340	3710	4078
Mean temperature (°C)	14.1	5.5	11	7.29
Mean annual rainfall (mm)	693	1157	747	609
Mean wind speed (m s^{-1})	3.65	2.2	5.43	3.61
Soil data				
Soil texture	Medium-Fine	Fine	Very fine	Medium
Soil depth (cm)	100	50	150	100
Agroforestry components				
Tree	*Quercus rotundifolia*	*Prunus avium*	*Populus spp*	*Populus* spp Max 1 var.
Crop	Non-improved Natural grasslands (ng)	Non-improved Natural grasslands (ng)	Wheat (w) Barley (b) Oilseed (o)	Sugar beet (sb) Wheat (w)
Crop rotation	ng	ng	w/w/b/o	sb/w/sb/w
Livestock	Iberian Pig/Cattle	Cattle	-	-

To assess how tree densities affected the supply of provisioning ecosystem services, six tree densities were analysed for each site. These included: 1) conventional agriculture with no trees (pasture-only or arable crop-only); 2) four agroforestry alternatives of different tree densities (labelled AF1, AF2, AF3 and AF4); and 3) tree-only systems with no crop or pasture production. The tree-only systems were developed assuming standard management practice for tree plantations at the locations assessed, and typically were established at a relatively high tree density, which was reduced over time using a thinning regime. The agroforestry alternatives were assumed to maintain the initial tree density throughout the simulation period (**Table 11**).

Table 11. Land use alternatives analysed in the study for the four agroforestry systems.

Land use alternative	***Montado***	**Cherry tree pastures**	**Silvo-arable systems**	**SRC**
Crop component Tree component	Natural grassland Holm oak	Natural grassland Cherry tree	Wheat (w) Barley (b) Oilseed (o) w/w/b/o rotation Poplar ssp	Sugar beet (sb) Wheat (w) sb/w/sb/w rotation Poplar spp
Monoculture	MONTPT-A	CTCH-A	SAFUK-A	SRCDE-A
Crop area (%)	100	100	100	100
AF1	MONTPT-AF1	CTCH-AF1	SAFUK-AF1	SRCDE-AF1
Crop area (%) Crop alley width (m) Tree density (ha^{-1}) Spacing between lines (m) Spacing within lines (m) Tree strip width (m)	99 - 50 5 14.1 Scattered trees	99 - 26 7 19.6 Scattered trees	80 10 39 10 18 3	94 96 497 96 0.9-1.8 11
AF2	MONTPT-AF2	CTCH-AF2	SAFUK-AF2	SRCDE-AF2
Crop area (%) Crop alley width (m) Tree density (ha-1) Spacing between lines (m) Spacing within lines (m) Tree strip width (m)	99 - 100 3.6 10 Scattered trees	99 - 52 5 13.6 Scattered trees	80 10 78 10 13 3	93 78 641 72 0.9-1.8 11
AF3	MONTPT-AF3	CTCH-AF3	SAFUK-AF3	SRCDE-AF3
Crop area (%) Crop alley width (m)	99 -	99 -	80 10	90 48

Tree density (ha-1)	150	78	117	905
Spacing between lines (m)	3	4	10	48
Spacing within lines (m)	8.1	11.3	9.5	0.9-1.8
Tree strip width (m)	Scattered trees	Scattered trees	3	11
AF4	MONTPT-AF4	CTCH-AF4	SAFUK-AF4	SRCDE-AF4
Crop area (%)	99	99	80	81
Crop alley width (m)	-	-	10	24
Tree density (ha^{-1})	200	104	156	1516
Spacing between lines (m)	2.5	3.5	10	24
Spacing within lines (m)	7	9.8	6.5	0.9-1.8
Tree strip width (m)	Scattered trees	Scattered trees	3	11
Forestry	MONTPT-F	CTCH-F	SAFUK-F	SRCDE-F
Crop area (%)	0	0	80	0
Initial tree density (ha^{-1})	505	690	1250	9672
Final tree density (ha^{-1})	100 (year 50)	100 (year 50)	158 (year 12)	7157 (year 3)
Tree strip width (m)	Scattered trees	Scattered trees	3	11

Iberian wood pastures

According to den Herder et al. (2017), the *montado* or *dehesa* systems in Portugal and Spain cover about 3.5 – 4.0 million hectares. The systems are characterized by low trees densities (20 to 50 trees ha^{-1}) combined with arable and/or pastoral activities. Depending on the main tree species present, two main different types of *montado* can be found: 1) cork oak *montado* where *Quercus suber* L. trees and cork extraction is the main economic activity, and 2) holm oak *montado* where *Quercus rotundifolia* L. is the dominant tree species, and the main economic activity is animal husbandry (cattle for beef production and/or Iberian pigs and also sheep/goat for meat, and milk derivates) under extensive practices. For this modelling assessment, the *montado* system was defined as a pure holm oak plantation, providing acorns between September and January, and grass during the entire year for grazing livestock. For the forestry system, typical thinning regimes were used. Livestock were absent but acorns were considered to be available as fruit from the trees. Regular pruning of the trees, removing 10% of the total biomass, was assumed to occur every 12 years to increase the light reaching the pasture (Olea and San Miguel-Ayanz 2006). Provisioning ecosystem services provided by the system included food in the form of meat from the livestock, and energy from the trees in the form of firewood derived from tree pruning and thinning. The livestock were considered to feed on pasture and acorns, when these were present.

Cherry orchards in Switzerland

Cherry tree orchards are traditional agroforestry systems that are widely spread throughout central Europe, and particularly in Switzerland (Sereke et al. 2015). These systems consist of tall, mixed fruit tree species, combined with grass or crops. Tree densities vary between 20 to 100 trees ha^{-1}, and the most common fruit tree species are apple (*Malus* spp.), pear (*Pyrus* spp.), plum (*Prunus domestica* L.), and cherry trees (*Prunus avium* L.). These species were primarily planted to provide fruit, but they also provide timber and nowadays the most common use of the wood is for firewood. The grass understorey was traditionally meadow or pasture for feeding animals. Despite a steady decline over recent years, these systems currently cover around 41,000 ha of agricultural land in Switzerland (Herzog 1998).

In this modelling assessment, the system was assumed to provide cherries during summer (June-July) and grass as fodder for cattle or sheep for the whole year. Pruning was assumed to occur every third year, and a one percent removal of the total biomass was also assumed to ensure constant fruit production. Timber was assumed to be used in furniture and was obtained in year 80 with the final harvest of the trees. For the agroforestry alternatives, it was considered that the management of the trees was for fruit production, whilst for the forestry system, the management was for timber production. The provisioning ecosystem services provided by cherry tree pastures thus included food in the form of cherries and grass for livestock grazing the pasture, material in the form of timber, and energy from wood for heating.

Silvo-arable systems in the UK

In 1992, a network of experimental silvo-arable systems was planted in the UK where rows of poplar trees (*Populus* spp) were planted with arable crops in alleys. As part of this, hybrid poplar for timber was planted in Silsoe (Bedfordshire, UK) in forestry and agroforestry schemes with cereal rotations including wheat (*Triticum* spp) and barley

(*Hordeum vulgare* L.). Arable control treatments were managed in the same way as the cereal intercrop and trees were planted at a tree density of 156 trees ha^{-1} with rows oriented in a north-south direction (Burgess et al. 2005).

Here, the combination of poplar trees for timber and cereal intercrops supplied food in the form of grain and material in the form of timber and cereal straw. The simulation period was 80 years and consisted of four poplar rotations of 20 years each. During tree growth, material from formation pruning was assumed to be discarded and was therefore not included in the analysis. In the agroforestry scenarios, tree density was increased, by reducing the distance between trees in the tree line so that the crop area remained constant at 80% of the total area.

Short rotation coppice in Germany

Short-rotation coppice (SRC) with poplar or other fast-growing species for the production of bioenergy is gaining interest as a possible means of decarbonising energy supplies. In temperate zones these systems can be used to produce biomass feedstocks, providing one approach to meeting increasing demand for self-sufficient energy supplies in decentralized rural areas (Gruenewald et al. 2007).

An agroforestry alley cropping trial was established in Forst (Lausitz, north-eastern Germany) in 2010 and 2011. The system included 11 m wide hedgerows with crop alleys ranging from 24 m to 96 m in width. The tree hedgerows included two poplar varieties, Max 1 (*Populus nigra* × *Poplar maximowiczii*) and Fritzi-Pauley (*Poplar trichocarpa*), and black locust (*Robinia pseudoacacia* L.). The trial area occupied around 40 hectares and tree densities in the tree rows were between 8,715 trees ha^{-1} and 9,804 trees ha^{-1} depending on whether a single or double row design was used.

Here, the modelling work assessed an alley cropping system with poplar Max 1 variety (*Populus nigra* × *Poplar maximowiczii*) SRC as the tree hedgerow, and winter wheat (*Triticum durum* L.) followed by sugar-beet, as the crop rotation. Tree coppicing was assumed to occur every four years, resulting in 20 rotations over the 80-year time horizon, and the trees were assumed to be replanted every five rotations. Due to the width of the tree strips in these systems, it was assumed that the two double rows located in the middle would behave like pure SRC, whilst the two double rows located on the edge of the tree strip would interact with the crop (Salkanovic 2017). The provisioning ecosystem services includes the supply of food from cereal grain and sugar-beet root, material from wheat straw, and the energy provided by the tree component.

Methodological approach for provisioning ecosystem services estimation

Yield-SAFE was used to predict the quantity of food, material, and energy provided by the trees, pasture, and crops for human and livestock consumption. Yield-SAFE was selected for this study for two reasons. Firstly, because it can model water and light capture and competition between tree and crop/pasture components in agroforestry systems, and secondly, because it can also simulate pure crop-only and pasture-only systems as well as tree-only systems, so that comparisons between the different land uses can be made.

For the calibration, a default set of parameters for the "potential" monoculture yields of trees, crops and pasture species was developed assuming no water limitation on growth yields (see full description in Graves et al. 2007. The model was then calibrated for site specific "reference" monoculture yields of the trees, crops, and pasture species. For this, crop and tree calibration data were taken from a variety of sources including Cubera et al (2009) and Palma et al. (2017c) for pasture and holm oak in Portugal, Graves et al. (2010) for crops and poplar yields in the UK, Palma et al. (2017c and Sereke et al. (2015) for pasture and cherry tree yields in Switzerland, and Mirck et al. (2016) for crop and poplar SRC in Germany (see **Figure 3**).

Weather data (daily solar radiation, temperature, and rainfall) were then extracted for use in Yield-SAFE from the CliPick tool (Palma 2017a) while information relating to soil texture and depth was provided by field data from the case study sites. Then selected parameters in Yield-SAFE (the water use efficiency, harvest index, and light use efficiency) were used to determine water limited reference yields for tree and crop species by adjusting these parameters within the ranges found in the literature so that estimated yields from Yield-SAFE matched the reference yields (see Graves et al., 2007 for full explanation).

Food production for human and livestock consumption was predicted by Yield-SAFE by estimating fruit yield using the methodology developed in Palma et al. (2016) that considers a fruit productivity per unit of tree leaf canopy parameter, the canopy cover and the tree density, for crops by estimating the grain or root yield, and for pasture by estimating the total pasture yield less 10% that was assumed to be left in the field after grazing. A livestock carrying capacity was quantified using data on the available UME of food (grass and/or acorns) consumed by the livestock and the livestock unit energy requirement (LUER: 103.2 MJ d^{-1}) as proposed by Hodgson (1990).

Raw materials were assumed to be outputs that would be used for on-farm construction or saleable products such as timber, bark, or straw. In this study, it was assumed that poplars from the UK, and cherry trees in Switzerland provided timber, estimated using the cumulative above ground biomass in year 80. For the *montado* system, the accumulated energy within the standing trees timber was included to complete the energy balance of the system, even though these trees would not normally be felled during an 80-year time horizon. Cereal straw was considered to be a material.

Energy was assumed to be produced either directly as the main output of the system (e.g. dedicated bioenergy plantations) or, indirectly as a by-product (e.g. pruning or thinning) from the tree component. For the alley cropping SRC system in Germany, energy production was viewed as the main output, and the management of the system was assumed to maximize fuelwood supply for local combined heat and power plants. For the *montado* system in Portugal, cherry orchard system in Switzerland, and poplar system in the UK, energy production was viewed as the consequence of management operations related to other objectives of the system, such as food or timber production, which nevertheless, provided an important food or fuel resource for local people.

Results and discussion

Modelled tree, crop, and pasture yields

The modelled yields of the crop-only or pasture-only systems for the simulation period were made to match observed reference values for each location and modelled variations in yields between years were due to differences in annual weather data, but were within the yield ranges reported for each case study area (**Figure 3**). The timber yields for trees were then compared with tree growth data derived for widely spaced oak trees (20 and 50 trees ha^{-1}) in *montado* in Portugal, cherry and poplar forests in Switzerland and the UK, and SRC agroforestry systems in Germany (**Figure 3**).

Under *montado,* at a density of 50 trees ha^{-1}, the simulated trees reached a height of 7 m, a diameter at breast height (DBH) of 40 cm, and an above-ground dry biomass of 570 kg, in year 80. These results seem reasonable as observed trees of the same age, range between 500 - 600 kg in above-ground biomass, 30-40 cm in DBH, and 5.5 - 8.0 m in height (Palma et al. 2017c). For the understorey component, the mean dry mass yield predicted by Yield-SAFE for monoculture pasture was 2.3 Mg ha^{-1} yr^{-1}. This value is within the range of 2.0 – 4.0 Mg ha^{-1} yr^{-1} reported in Cubera et al. (2009).

In Switzerland, timber production results from Yield-SAFE were similar to published data. For year 60, Yield-SAFE estimated a timber volume of 1.4 m3 $tree^{-1}$ and 0.99 m3 $tree^{-1}$ respectively for 40 trees ha^{-1} in the agroforestry system and for the tree-only system. These results are in accordance with Sereke et al. (2015) who for year 60 reported timber volumes of 1.34 m3 $tree^{-1}$, 1.14 m3 $tree^{-1}$, and 1.07 m3 $tree^{-1}$ respectively, for two wild cherry timber in Switzerland at 40 and 70 trees ha^{-1} (both agroforestry systems), and a forestry system, with an establishment density of 816 trees ha^{-1}, thinned to a final density of 100 trees ha^{-1} in year 60.

For the silvo-arable system in the UK, simulation results using Yield-SAFE were coherent with the results obtained at experimental sites in the UK. Graves et al (2010) reported yields for winter wheat, barley, and oilseed rape of 8.23 Mg ha^{-1}, 6.83 Mg ha^{-1} and 3.44 Mg ha^{-1} respectively. The simulated results for the initial years of the agroforestry system achieved similar results on a per hectare crop basis for winter wheat, barley, and oilseed of 8.30 Mg ha^{-1}, 6.92 Mg ha^{-1} and 3.30 Mg ha^{-1} respectively. But these declined as the trees started to grow. Graves et al (2010) used timber volumes per tree of 0.35 and 2.41 m^3 in year 12 and 30 respectively to calibrate Yield-SAFE for the forestry reference system. Here, simulations over a 30-year time horizon predicted timber yields of 1.9 m^3 $tree^{-1}$ and 2.7 m^3 $tree^{-1}$ for the forestry and for the agroforestry alternative (156 trees ha^{-1}) respectively.

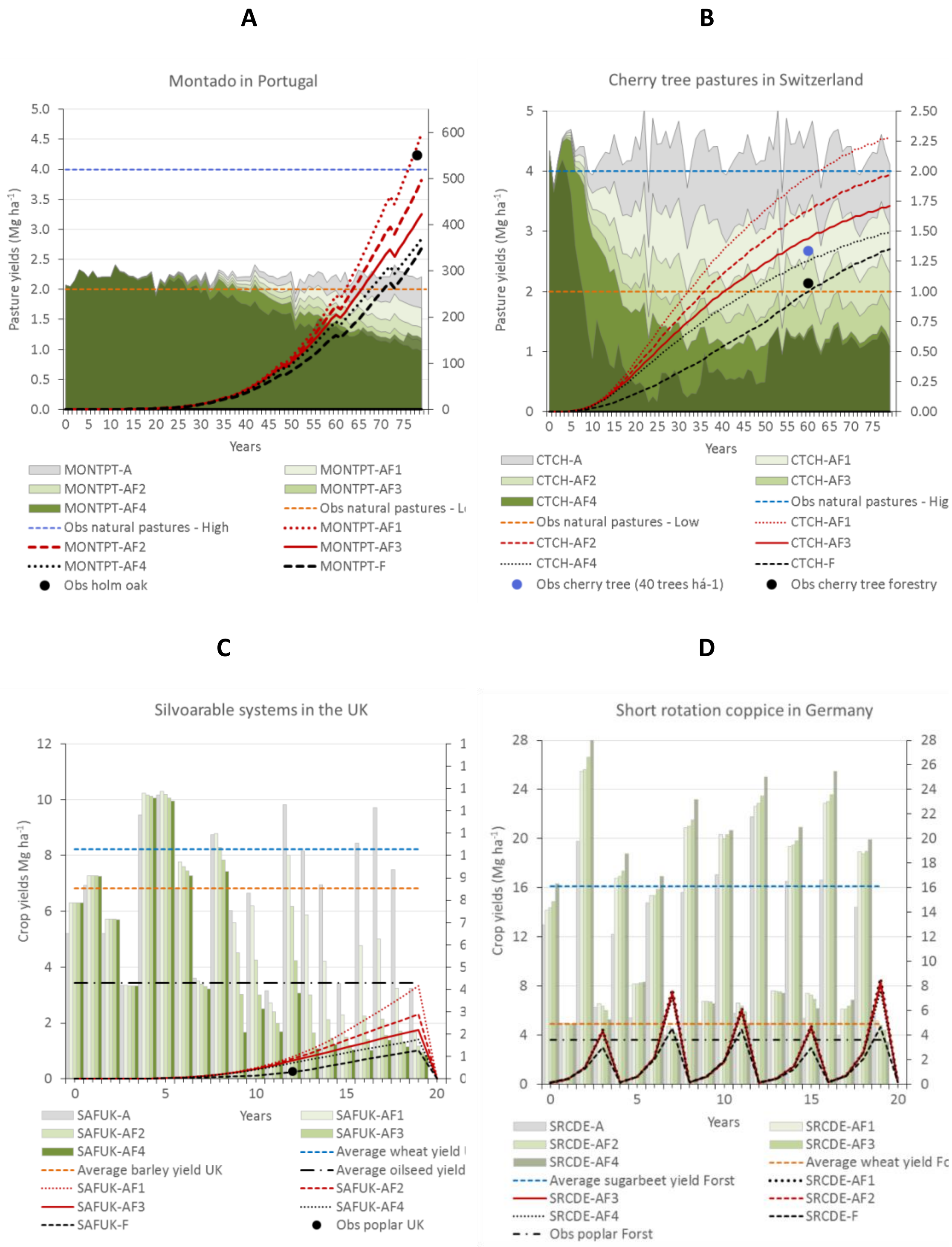

Figure 3. Tree growth and crop yield validation for six different management alternatives in increasing tree densities across Europe. For the systems in Portugal (A) and Switzerland (B) a simulation of period of 80 years is shown. For the systems in the UK (C) and the SRC in Germany (D) a simulation period of 20 years is shown.

A: *Montado* systems in Portugal: 0 trees ha^{-1} (MONTPT-A); 50 trees ha^{-1} (MONTPT-AF1); 100 trees ha^{-1} (MONTPT-AF2); 150 trees ha^{-1} (MONTPT-AF3); 200 trees ha^{-1} (MONTPT-AF4) and forestry (MONTPT-F). Observed yields in natural pastures range from low (2 Mg ha^{-1}) to high (4 Mg ha^{-1}) as reported by Cubera et al (2009). The observed holm oak yield of 550 kg tree^{-1} is reported by Palma et al (2017c).

B: Cherry tree systems in Switzerland: 0 trees ha^{-1} (CTCH-A); 26 trees ha^{-1} (CTCH-AF1); 52 trees ha^{-1} (CTCH-AF2); 78 trees ha^{-1} (CTCH-AF3); 104 trees ha^{-1} (CTCH-AF4) and forestry (CTCH-F). Observed natural pasture yields range from low (2 Mg ha^{-1}) to high (4 Mg ha^{-1}) as reported by Agroscope in 2015. Observed agroforestry cherry trees (40 tree ha^{-1}) (1.34 m^3 $tree^{-1}$) and cherry forestry (1.14 m^3 $tree^{-1}$) tree volumes are reported in Sereke et al (2015).

C: Poplar systems in the UK: 0 trees ha^{-1} (SAFUK-A); 39 trees ha^{-1} (SAFUK-AF1); 78 trees ha^{-1} (SAFUK-AF2); 117 trees ha^{-1} (SAFUK-AF3); 156 trees ha^{-1} (SAFUK-AF4) and forestry (SAFUK-F). Average wheat (8.23 Mg ha^{-1}), barley (6.83 Mg ha^{-1}) and oilseed (3.44 Mg ha^{-1}) yields in arable system the UK and poplar tree volume in year 12 (Obs poplar UK) of 0.35 m^3 $tree^{-1}$ for a 156 tree ha^{-1} forestry system are reported in Graves et al (2010).

D: Short rotation coppice (SRC) in Germany: pure arable (SRCDE-A); alley widths of 96 m (SRCDE-AF1); 72 m (SRCDE-AF2); 48 m (SRCDE-AF3); 24 m (SRCDE-AF4) and pure SRC (SRCDE-F). Average yields of winter wheat (4.9 Mg ha^{-1}) and sugar beet (16.1 Mg ha^{-1}) reported for the Forst site by Mirck (2016). Observed poplar for Forst corresponds to the average tree biomass of poplar of 3.59 kg $tree^{-1}$ in an SRC system with an initial tree density of 8,497 trees ha^{-1} finishing at 6,295 tree ha^{-1}.

In Germany arable monoculture simulations using Yield-SAFE provided similar crop yields (4.0 - 4.5 Mg ha^{-1} for winter wheat and 16.52 Mg ha^{-1} for sugar beet, both in dry weight) to the results obtained for the Forst site control plot (4.9 Mg ha^{-1} and 16.1 Mg ha^{-1} respectively). The tree dry biomass results from Yield-SAFE (3.56 kg $tree^{-1}$) were similar to those reported by Mirck (2016) and Kanzler and Mirck (2017) after four years for a stand density starting at 8,497 trees ha^{-1} and finishing at 6,295 tree ha^{-1} to give a total stand biomass yield of 22.6 Mg ha^{-1} (3.59 kg $tree^{-1}$).

Predicted supply of provisioning ecosystem services

The systems showed substantial differences in the energy accumulated over the simulation period (**Figure 4**). There was a large difference between the *montado* that was only able to accumulate between 2.8 and 4.8 million MJ ha^{-1} over 80 years and the silvo-arable systems in the UK which accumulated between 12 and 17 million MJ ha^{-1} over 80 years. Between these extremes, the Swiss cherry tree systems and the SRC systems in Germany were able to accumulate between 5 and 10 million MJ ha^{-1} and 10 and 14 million MJ ha^{-}1 over 80 years respectively. These results were largely explained by 1) differences in weather and soil conditions between the four biogeographical regions that influenced the potential biomass growth of the systems, and 2) differences in the light and water use efficiency of the tree and crop elements that formed each system.

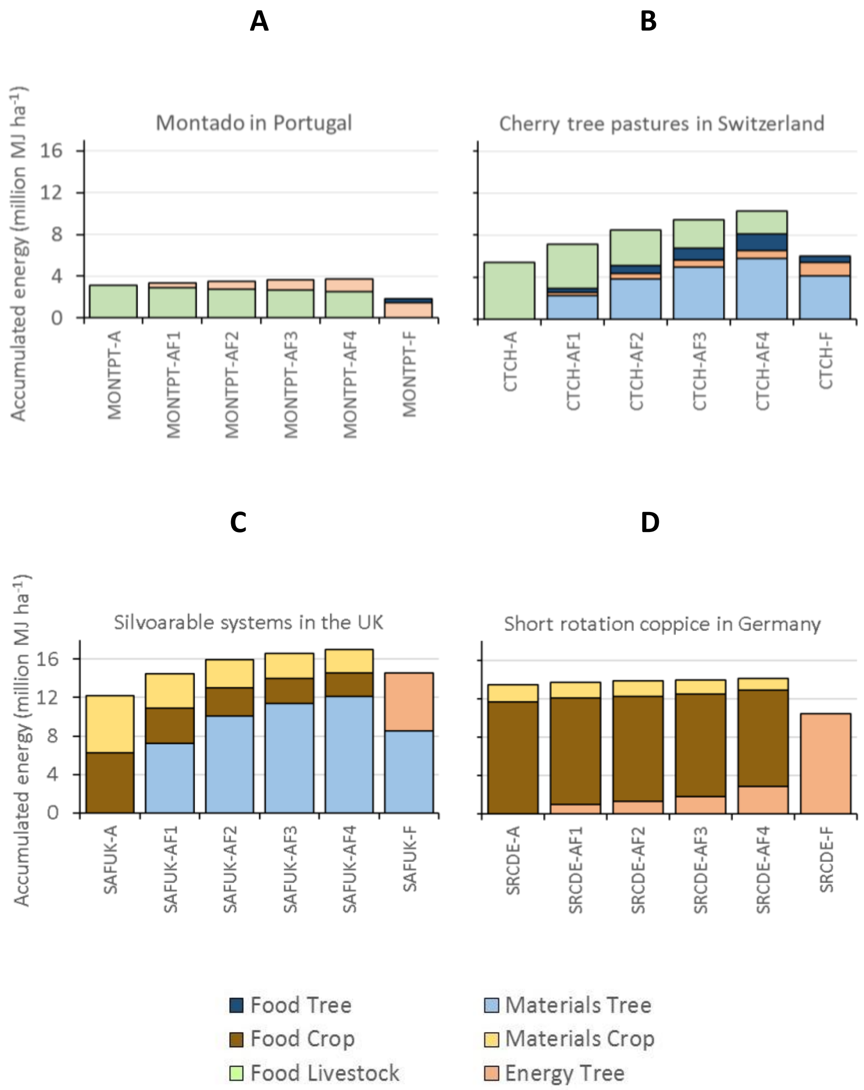

Figure 4. Accumulated energy of provisioning ecosystem services over 80 years for six different management alternatives at increasing tree densities across Europe.

A: *Montado* in Portugal: 0 trees ha^{-1} (MONTPT-A); 50 trees ha^{-1} (MONTPT-AF1); 100 trees ha^{-1} (MONTPT-AF2); 150 trees ha^{-1} (MONTPT-AF3); 200 trees ha^{-1} (MONTPT-AF4) and forestry (MONTPT-F).

B: Cherry tree pastures in Switzerland: 0 trees ha^{-1} (CTCH-A); 26 trees ha^{-1} (CTCH-AF1); 52 trees ha^{-1} (CTCH-AF2); 78 trees ha^{-1} (CTCH-AF3); 104 trees ha^{-1} (CTCH-AF4) and forestry (CTCH-F).

C: Silvo-arable systems in the UK: 0 trees ha^{-1} (SAFUK-A); 39 trees ha^{-1} (SAFUK-AF1); 78 trees ha^{-1} (SAFUK-AF2); 117 trees ha^{-1} (SAFUK-AF3); 156 trees ha^{-1} (SAFUK-AF4) and forestry (SAFUK-F).

D: Short rotation coppice (SRC) in Germany (D): pure arable (SRCDE-A); 96 m alley width (SRCDE-AF1); 72 m (SRCDE-AF2); 48 m (SRCDE-AF3); 24 m (SRCDE-AF4) and pure SRC (SRCDE-F).

The *montado* is an extensive silvo-pastoral system where livestock feed mainly on pasture, and acorns provide additional energy. Modelling results suggested that there was a decrease in food for livestock as tree canopy area increased over the simulation period (**Figure 5**). In the agroforestry systems, this decrease in the energy available for livestock food was associated with a reduction in pasture yield due to tree competition for water and light of 12% and 50% in year 80 for the 50 and 200 trees ha^{-1} densities respectively, which was not compensated for by the availability of acorns. In the forestry system, the additional energy provided by the thinnings and the acorns also failed to compensate for the loss of the energy that was available in the grass as grass was assumed not to grow in the system. Thus, the overall energy accumulated by the forestry system was lower than that accumulated in the treeless pasture and agroforestry systems (**Figure 4** and **Figure 5**).

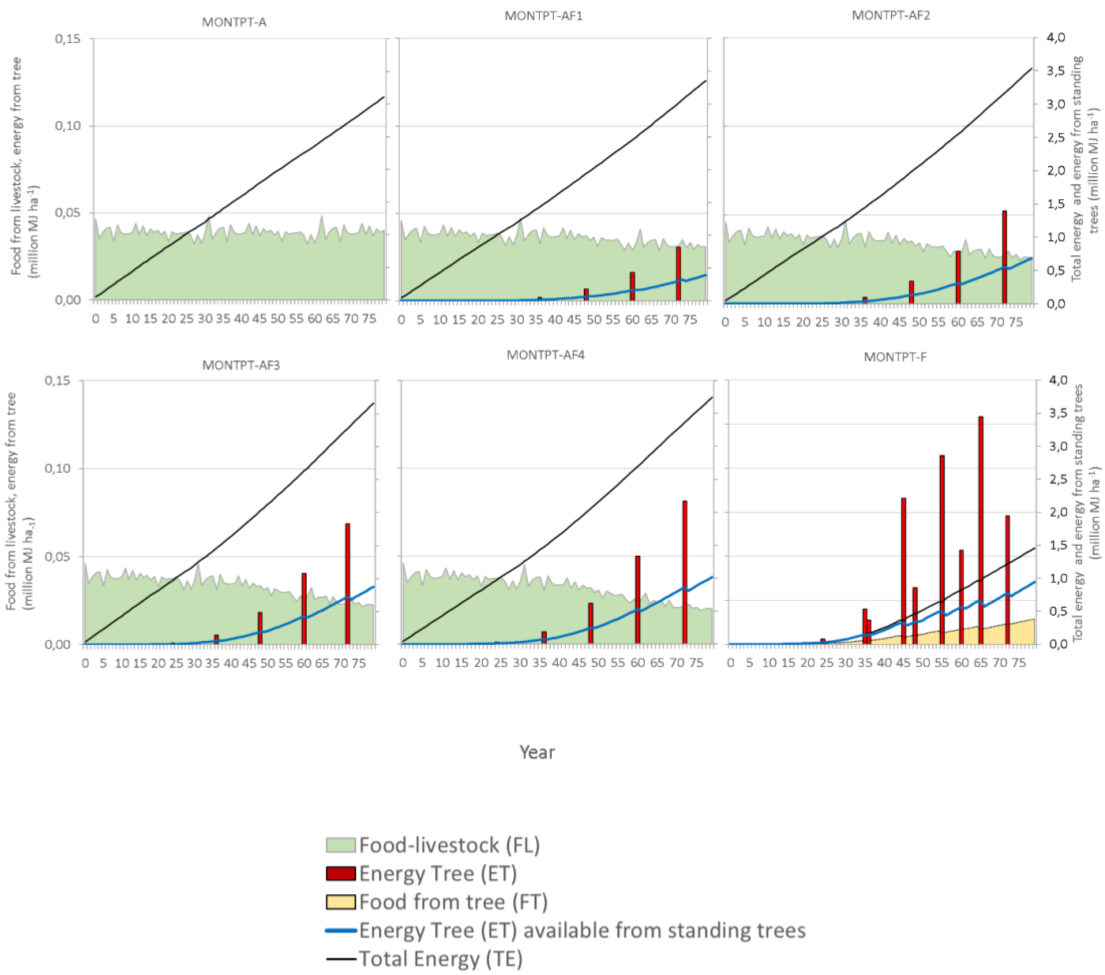

Figure 5. Annual energy of the food available for livestock (FL), the energy from tree pruning and thinning, the available energy from the standing biomass (left-hand scale) and the accumulated energy (right-hand scale) over 80 years for six *montado* tree densities: 0 trees ha^{-1} (MONTPT-A); 50 trees ha^{-1} (MONTPT-AF1); 100 trees ha^{-1} (MONTPT-AF2); 150 trees ha^{-1} (MONTPT-AF3); 200 trees ha^{-1} (MONTPT-AF4) and forestry (MONTPT-F).

The modelled results for year 80 showed carrying capacities of 0.9 LU ha^{-1} (93 MJ d^{-1}) and 0.7 LU ha^{-1} (72 MJ d^{-1}) for the agroforestry system at 50 trees ha^{-1} (MONTPT-AF1) and 100 trees ha^{-1} (MONTPT-AF2) respectively (**Figure 6**). These values were higher than the optimum livestock carrying capacity of 0.18 - 0.60 LU ha^{-1} (62 MJ d^{-1}) reported by Godinho et al. (2014) for mature a *montado* with a canopy cover of 20 - 50% in southern Portugal. This may be because farmers must consider stocking levels in the context of other management factors, for example, the periods of low grass production, and the cost of feed imported or produced in more productive areas of the farm to support livestock between grass production peaks (Moreno et al. 2018). The simulations (**Figure 6**) suggested that only pure pasture (MONTPT-A) maintained daily energy values above the LUER threshold of 103 MJ d^{-1}. All alternative tree-based systems showed lower food energy availability with the decrease being greater as tree density increased (**Figure 5**). It is worth noting that during the first 30 years, pasture yields in the agroforestry systems were similar to the yields in the tree-less pasture. During this stage, the impact of the trees in terms of light and water competition was relatively small (**Figure 6**).

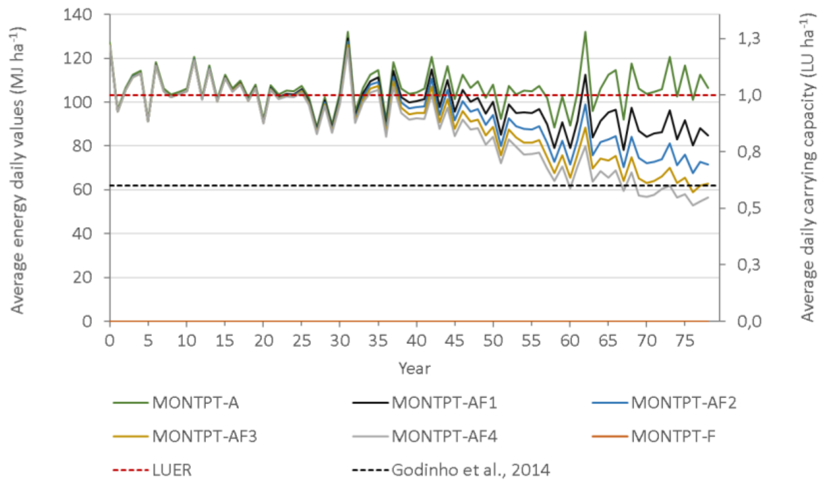

Figure 6. Mean daily values of food provided for livestock (MJ ha^{-1}) for the six *montado* management alternatives with a reference value of 103.2 MJ d^{-1} (daily livestock unit energy requirement - LUER) and the maximum carrying capacity suggested by Godinho et al. (2014) of 61.9 MJ d^{-1} (60% of LUER, dotted black line). The six *montado* alternatives are: 0 trees ha^{-1} (MONTPT-A); 50 trees ha^{-1} (MONTPT-AF1); 100 trees ha^{-1} (MONTPT-AF2); 150 trees ha^{-1} (MONTPT-AF3); 200 trees ha^{-1} (MONTPT-AF4) and forestry (MONTPT-F).

In the Swiss agroforestry systems, the energy accumulation in the fruit production of the trees increased with tree density (**Figure 4**) and during the first 25 years (**Figure 7 and Figure 8A**). Fruit production was lowest in the forestry system despite the initial high density at planting, because the trees were managed for timber production rather than fruit production. The Swiss forestry system (CTCH-F) showed no energy accumulation in grass (**Figure 8B**) since there was no predicted grass production. In the agroforestry alternatives, the accumulated energy in grass increased as tree density decreased. The energy accumulation in grass was predicted to endure over the whole rotation, although at reduced levels in comparison with the pasture only system **(Figure 6B).** Whilst the pure pasture (CTCH-A) system maintained energy values at levels that were able to support approximately 1.5 LU ha^{-1} indefinitely, the energy accumulated in the grass of the agroforestry alternative with the lowest tree density (CTCH-AF1) was the only one able to maintain 1.0 LU ha-^{1} until year 80.

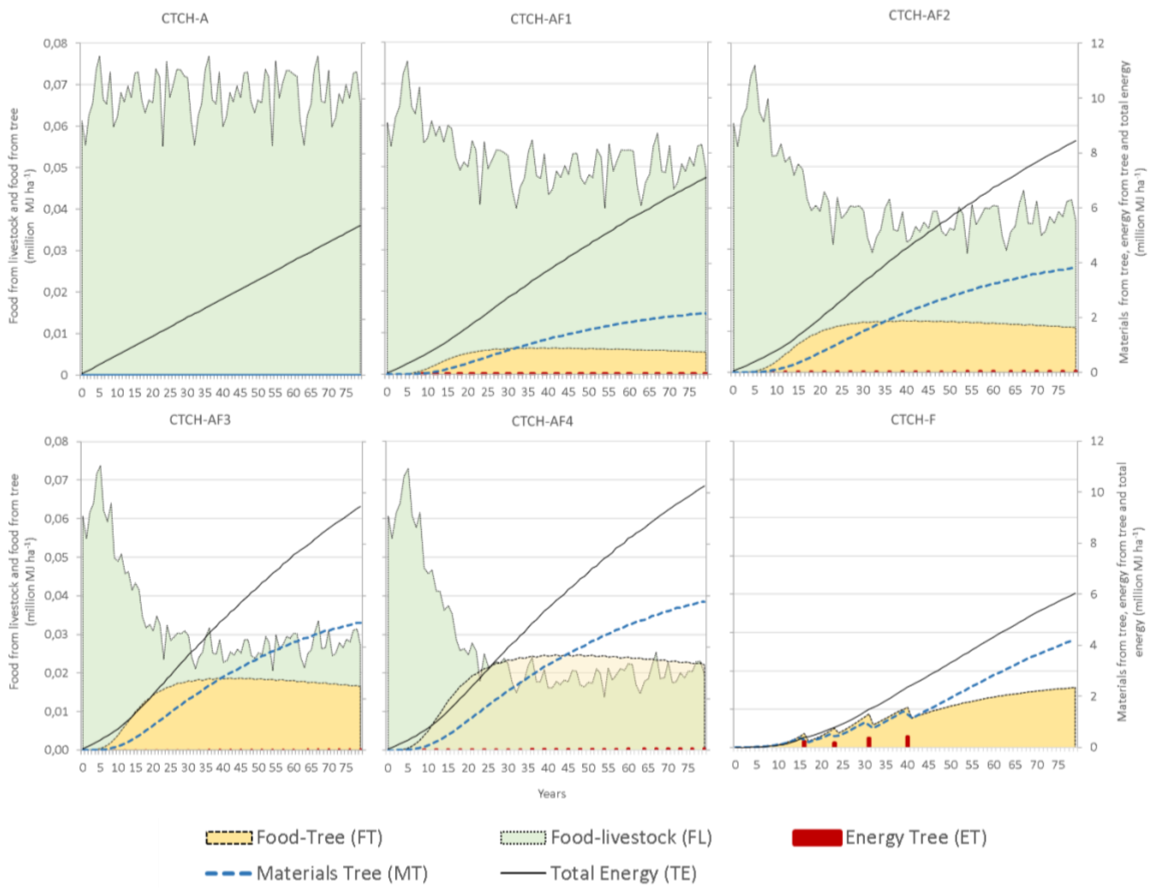

Figure 7. Annual value of the energy of food provided by the trees (FT), of food provided for livestock (FL), and energy provided by trees (ET) (left-hand scale), and the accumulated value of the energy in materials in the trees (MT) and the total energy (TE) of the system (MJ ha^{-1}) (right-hand scale) for cherry tree pastures in Switzerland with six different tree densities: 0 trees ha^{-1} (CTCH-A); 26 trees ha^{-1} (CTCH-AF1); 52 trees ha^{-1} (CTCH-AF2); 78 trees ha^{-1} (CTCH-AF3); 104 trees ha^{-1} (CTCH-AF4) and Forestry (CTCH-F).

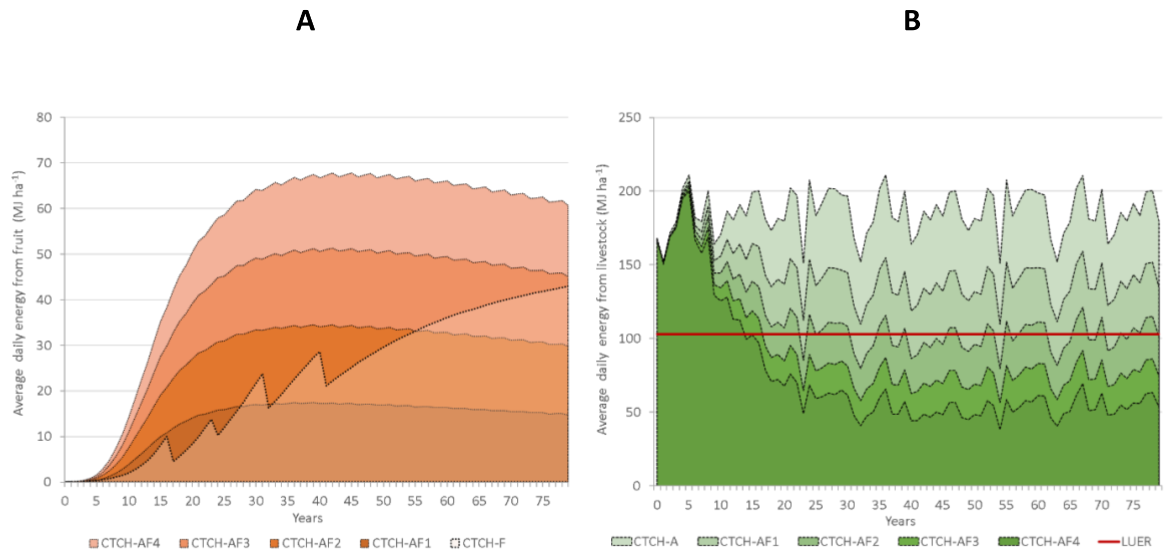

Figure 8. Mean annual values of the daily energy provided A) by fruit from trees (FT) and B) food provided for livestock (FL) over 80 years from the six systems analysed in Switzerland: 0 trees ha^{-1} (CTCH-A); 26 trees ha^{-1} (CTCH-AF1); 52 trees ha^{-1} (CTCH-AF2); 78 trees ha^{-1} (CTCH-AF3); 104 trees ha^{-1} (CTCH-AF4) and tree-only system (CTCH-F).

In the silvo-arable systems in the UK, the energy accumulated by the trees compensated for the energy lost as crop production decreased due to the competitive effect of the trees (**Figure 4** and **Figure 9**). The greatest energy accumulation occurred in the most densely planted agroforestry system and the lowest in the arable system. The forestry system (SAFUK-F) despite being planted at a high density and thinned, provided an intermediate level of energy accumulation over the simulation period that was higher than the crop only system, but lower than the all but the most widely spaced agroforestry system.

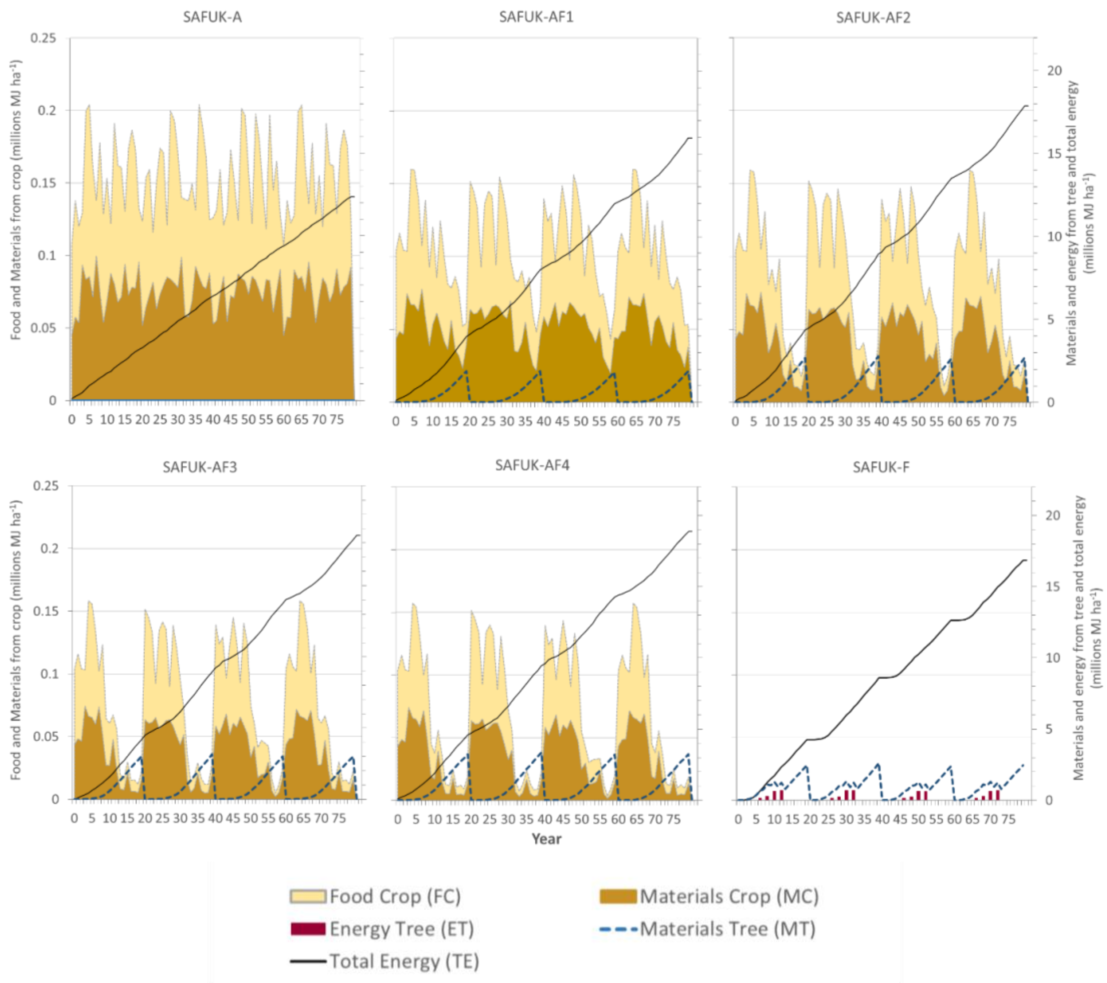

Figure 9. Annual energy value of food (FC) and materials (MC) provided by crops (left-hand scale), and the annual energy provided from tree thinning (ET), and the growth of the trees (MT), and the accumulated total energy (TE) (right-hand scale) for silvo-arable systems in the UK over 80 years for six different tree densities: 0 trees ha^{-1} (SAFUK-A); 39 trees ha^{-1} (SAFUK-AF1); 78 trees ha^{-1} (SAFUK-AF2); 117 trees ha^{-1} (SAFUK-AF3); 156 trees ha^{-1} (SAFUK-AF4) and Forestry (SAFUK-F).

In Germany, in the agroforestry systems, the hedgerows of poplar SRC increased the total energy accumulated by the system as the crop alley decreased in width (**Figure 10**). The energy accumulated in the most densely planted agroforestry system was greatest out of all the systems. The crop only system accumulated marginally lower levels of energy than the most widely planted agroforestry system. The forestry system accumulated the lowest quantity of energy out of all the systems.

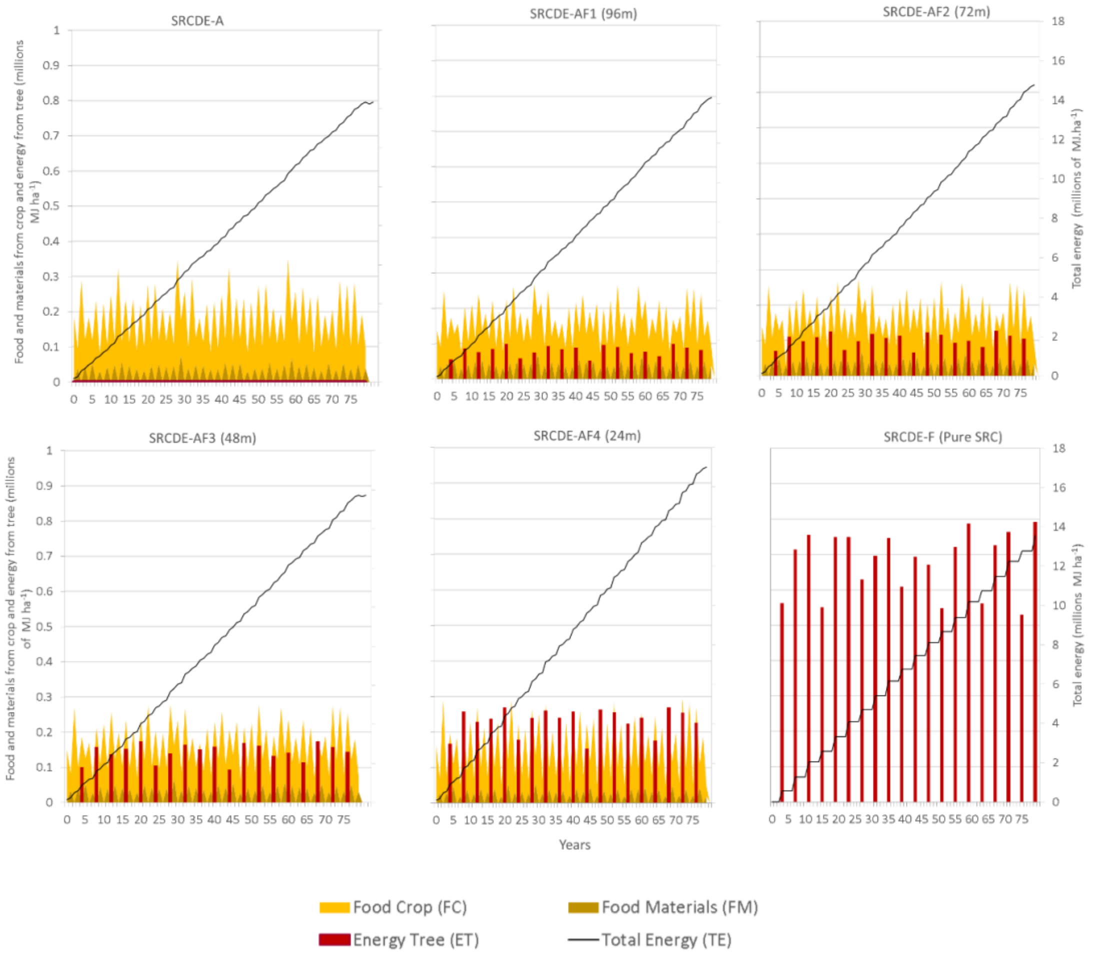

Figure 10. Annual energy value of food (FC) and the material (MC) provided by the arable crop, and the energy provided by trees (ET) (left-hand scale) and the accumulated total energy (TE) by the system over 80 years for a short rotation system (SRC) in Germany with six different crop alley widths: no SRC (SRCDE-A); 96 m (SRCDE-AF1); 72 m (SRCDE-AF2); 48 m (SRCDE-AF3); 24 m (SRCDE-AF4) and pure SRC (SRCDE-F).

Tree density effects on energy accumulated in provisioning ecosystem services

All the agroforestry systems showed an increase in accumulated energy as tree density increased with the highest tree density system (AF4) (**Figure 4**) showing the greatest accumulated energy at the four locations. This density was 200 trees ha^{-1}, 104 trees ha^{-1}, 156 trees ha^{-1}, and 24 m alley widths for the Portuguese *montado*, Swiss cherry tree pastures, UK poplar silvo-arable systems and the German SRC respectively. This increase in accumulated energy was relatively small for the *montado* systems in Portugal and the SRC systems in Germany and relatively large for the poplar silvo-arable system in the UK and the Swiss cherry pasture systems. Conversely, the energy accumulated in provisioning ecosystem services was lowest for one of the monoculture systems at each site. This was for the forestry system in Portugal, pasture system in Switzerland, crop system in the UK, and pure SRC system in Germany.

However, the energy accumulated per tree also varied as tree density changed. For all the systems, the energy accumulated per tree was greatest in the AF1 systems which had the lowest tree densities (**Figure 11**). This was 50 trees ha^{-1}; 26 trees ha^{-1}; 56 trees ha^{-1} and 96 m alley widths for the Portuguese *montado*, Swiss cherry tree pastures, UK poplar silvo-arable systems and the German SRC respectively.

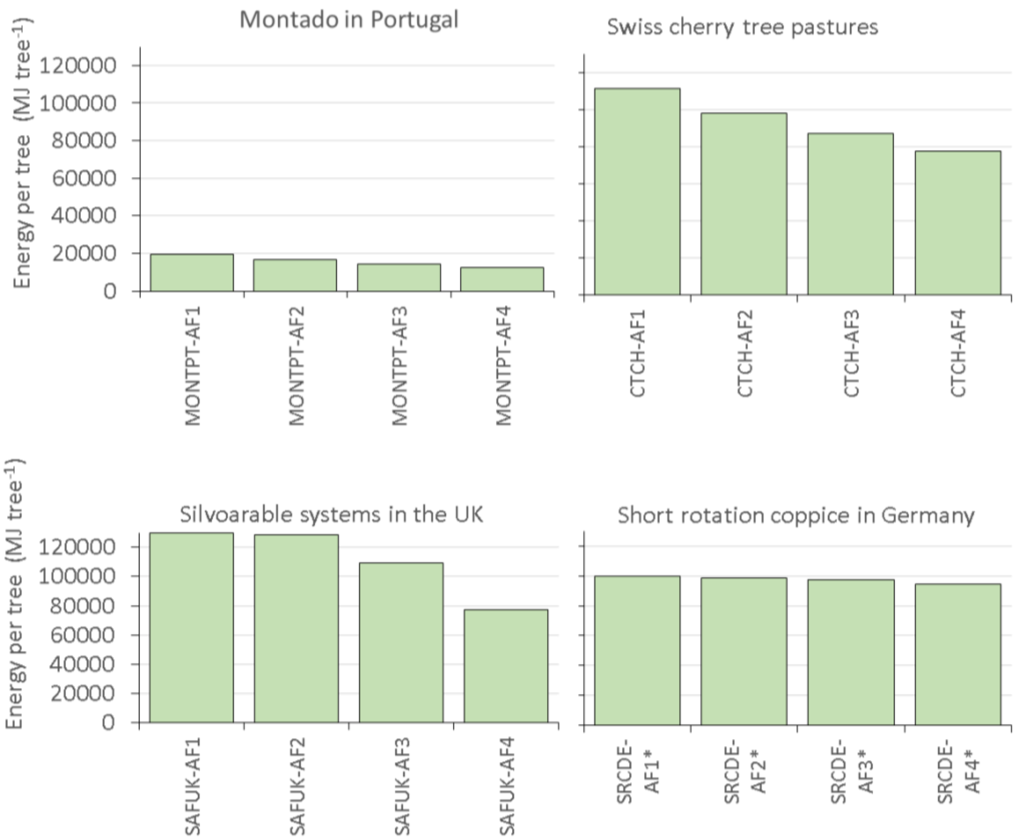

Figure 11. Energy accumulation per tree over 80 years for the four-case study agroforestry systems Europe. These are:

A: *Montado* in Portugal: 50 trees ha-1 (MONTPT-AF1); 100 trees ha-1 (MONTPT-AF2); 150 trees ha-1 (MONTPT-AF3); 200 trees ha-1 (MONTPT-AF4).

B: Cherry tree pastures in Switzerland: 26 trees ha-1 (CTCH-AF1); 52 trees ha-1 (CTCH-AF2); 78 trees ha-1 (CTCH-AF3); 104 trees ha-1 (CTCH-AF4).

C: Silvo-arable systems in the UK: 39 trees ha-1 (SAFUK-AF1); 78 trees ha-1 (SAFUK-AF2); 117 trees ha-1 (SAFUK-AF3); 156 trees ha-1 (SAFUK-AF4).

D: Short rotation coppice (SRC) in Germany (D): 96 m alley width (SRCDE-AF1); 72 m (SRCDE-AF2); 48 m (SRCDE-AF3); 24 m (SRCDE-AF4). Values for this system are per hundred trees.

*values per 50 trees

As reported by Graves et al. (2007; 2010), an increase in tree density in agroforestry systems increases inter-tree competition and tree-crop competition for light, water, and nutrients and this leads to a reduction of the overall biomass produced per tree. Although trees planted at high density in the field may increase the capture of water with deeper roots, or the capture of radiation by increasing canopy height or leaf area (Eastham et al. 1990; Toillon et al. 2013), or benefit from mutual shelter from the wind, such effects were not included in Yield-SAFE.

A key factor in considering the effects of tree density on the accumulated energy in provisioning ecosystem services is in terms of the tree canopy effects on resource availability and how this affects tree, pasture, crop and livestock productivity. Further research is needed to allow Yield-SAFE to account for microclimatic effects. These should include the tree canopy effects on wind, temperature, and vapour pressure deficits (Palma et al 2017c) as observations have shown that productivity in agroforestry systems can also be increased in part due to soil moisture benefits (Cubera et al. 2009; Rivest et al. 2011; López-Díaz et al. 2015). By considering only water and light competition, Yield-SAFE currently omits important benefits of the tree canopy, potentially under predicting the beneficial impacts this may have on livestock and understory pasture and crops. The results presented here may therefore be conservative and the energy accumulated in the provisioning ecosystem services under estimated. For example, in the case of livestock, tree canopies provide shade and shelter from extreme weather conditions, reducing the energy they metabolise for body temperature regulation and increasing the energy available for body weight gain. The effects trees can have on reducing wind speed, temperature, and vapour pressure deficits, conserves soil moisture and potentially extends the growth period for pasture and crops, allowing for greater energy accumulation within the system.

Conclusions

Agroforestry provides an approach for diversifying the supply of provisioning ecosystem services from the same land area whilst at the same time reducing the negative environmental impacts associated with agriculture. This paper provides a process-based understanding of the effects of tree density on four contrasting agroforestry systems in different parts of Europe. Using Yield-SAFE, the effects of different tree densities on the food, material, and energy production in terms of total energy production over an 80-year simulation period were predicted. These showed that adding trees to monoculture arable or pasture systems increased the accumulated energy, indicating resource-use complementarity between the different components. The accumulated energy varied from relatively low values in the drought stressed *montado* of Portugal and temperature-limited tree-pasture systems of Switzerland, to relatively high values in the temperate environments of the UK and Germany, demonstrating the importance of edapho-climatic conditions when assessing agroforestry systems. However, the increase in energy accumulated due to tree presence was not linear, and tree competition for water and solar radiation increased with tree density, reducing the quantity of understory biomass that could be produced, and the energy accumulated per tree.

Although there are limitations in using energy as a common measure, the approach presented here does provide a means of quantifying the production of provisioning ecosystem services across different tree densities and land use systems in different environmental conditions. The research could be taken forward in two principal ways. Firstly, the economic value of the different components could be considered. A second approach is to incorporate other important effects of tree canopies on understorey crop and pasture growth, such as temperature and wind speed effects, to study how these effects might be used to improve advice on agroforestry systems.

Acknowledgments

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Chapter 4| Quantifying regulating ecosystem services with increased tree densities on European farmland

Based on **Crous-Duran J**, Graves AR, García de Jalón S, Kay S, Tomé M, Burgess PJ, Giannitsopoulos M, Palma JHN. Quantifying regulating ecosystem services in increasing tree densities in European farmland. Sustainability 2020, 12, 6676; doi:10.3390/su12166676

Abstract

Agroforestry systems have been compared to agricultural and forestry alternatives, providing a land use solution for additional environmental benefits while maintaining similar levels of productivity. However, there is scarce research assessing such pattern across a pan-European scale, using a common methodology. This study aims to improve our understanding of the role of trees on three different Regulating Ecosystem Services – 1) soil erosion, 2) nitrate leaching and 3) carbon sequestration – in traditional and innovative agroforestry systems in Europe through a consistent modelling approach. The systems' assessment spans environmentally from Mediterranean in Portugal, Continental in Switzerland and Germany, to Atlantic in the United Kingdom. Modelled tree densities were compared in the different land-use alternatives ranging from zero (agriculture with only crops or pasture) to forestry (only trees). The methodology included the use of a bio-physical model (Yield-SAFE) where the quantification of the environmental benefits was integrated. Results show a consistent improvement of Regulating Ecosystem Services can be expected when introducing trees in the farming landscapes in different environmental regions in Europe. For all the systems, the forestry alternatives presented the best results in terms of decreasing in soil erosion of 51% (± 29), decreasing nearly all the nitrate leaching (98% ± 1) and an increasing of the carbon sequestration up to 238 Mg C ha^{-1} (± 140). However, these alternatives are limited in the variety of food, energy and/or materials provided. On the other hand, from an arable or pure pasture alternative starting point, an increase in agroforestry tree density could also be associated to a decrease on soil erosion up to 25% (±17), a decrease on nitrates leached up to 52% (±34) and an increase on the carbon sequestered of 163 Mg C ha^{-1} (± 128) while at the same time ensuring same levels of biomass growth and an increase in product diversification.

Keywords

Yield-SAFE, process-based model, agroforestry, carbon sequestration, soil erosion, nitrate leaching, *montado*, short rotation coppice, silvo-arable, fruit orchards, tree density

Introduction

Agroforestry systems (AFS) are attracting the interest of land managers who are seeking a more efficient way of producing food, bioenergy and materials than found with monoculture agriculture or forestry (Graves et al. 2007). This is because AFS can enable higher and more diverse biomass production due to the greater capture of solar radiation and water by trees and crops when both are grown together (Cannell et al. 1996; Crous-Duran et al. 2018).

At the same time, compared to agriculture, agroforestry can reduce soil erosion (Nair 2007), nitrate leaching (Jose 2009), and net greenhouse gas emissions (Godfray et al. 2012), whilst improving biodiversity (Klaa et al. 2005) and enhancing climate change mitigation by sequestering carbon (Cardinael et al. 2015). These sorts of ecosystem benefits that mediate or moderate the effect of the environment on human well-being and health are defined by the Common International Classification of Ecosystem Services (CICES, v5.1) as regulation and maintenance services.

Some studies have focused on the effect of a single agroforestry system on one ecosystem service for one location. However, there have been few studies including different ecosystems services across different sites which could help establish relationships at broader scales (Moreno et al. 2017). Because of the difficulty in establishing a full range of practices in a single location, many comparisons of agroforestry with agricultural systems rely on the use of models (Burgess and Rosati 2018).

In a previous paper a bio-physical model called Yield-SAFE (van der Werf et al. 2007), which can predict tree and arable crop growth and yields in agroforestry systems, was used to examine how different densities of trees affected the provisioning services of four contrasting locations and agroforestry systems across Europe. The paper concluded that including trees in pasture or arable systems increased the capture of solar energy and thereby the production of total biomass per unit area in comparison with monoculture forestry, pasture, and arable systems, but that the accumulated energy per tree was reduced as tree density increased (Crous-Duran et al. 2018). The aim of this paper is to determine how an increase in tree density and the associated tree-crop interactions for water and radiation interact with the supply of three different Regulating Ecosystem Services (RES): regulation of soil erosion, regulation of nitrate leaching, and carbon sequestration, with a consistent methodology across Mediterranean, Continental and Atlantic environmental regions in Europe.

Materials & Methods

Agroforestry case study systems

Four agroforestry systems were considered for this study: 1) Iberian wood pastures called *dehesa* in Spain and *montado* in Portugal with holm oak (MONTPT); 2) grazed cherry orchards in Switzerland (CTCH); 3) poplar silvo-arable systems in the United Kingdom (SAFUK) and 4) fast-growing poplar short rotation coppice plantations for energy purposes within arable fields in Germany (SRCDE) (**Table 12**).

Table 12. Location, meteorological and soil information and component description of the four agroforestry systems studied.

	Montado	**Cherry orchards**	**Silvo-arable systems**	**SRC**
Location	Montemor-o-Novo, Portugal	Gempen, Switzerland	Silsoe, United Kingdom	Forst, Germany
Identification	MONTPT	CTCH	SAFUK	SRCDE
Altitude (m asl)	130	680	70	75
Longitude (°)	38.7023	7.2299	50.0089	51.7890
Latitude (°)	-8.3261	6.9943	0.4358	14.4918
Meteorological conditions				
Mean annual solar radiation (MJ m^{-2})	6080	4340	3710	4078
Mean annual temperature (°C)	14.1	5.5	11	7.29
Mean annual rainfall (mm)	693	1157	747	609
Mean wind speed (m s^{-1})	3.65	2.2	5.43	3.61
Soil data				
Soil texture	Medium-Fine	Fine	Very fine	Medium
Soil depth (cm)	100	50	150	100
Agroforestry components				
Tree	*Q. rotundifolia*	*Prunus avium*	Populus spp	Populus spp
Crop	Non-improved Natural grasslands (ng)	Non-improved Natural grasslands (ng)	Wheat (w) Barley (b) Oilseed (o)	Sugar beet (sb) Wheat (w)
Crop rotation	none	none	w/w/b/o	sb/w/sb/w
Livestock	Iberian Pig/Cattle	Cattle	-	-

These systems 1) represent traditional and innovative agroforestry systems in different climatic conditions, 2) focus on different products (tree, livestock and/or crop components) and 3) there are long-term experimental trials with data available for validation. Each agroforestry system was considered in terms of four agroforestry alternatives (different densities depending on each system), an agriculture (no trees) and a high-density tree-only system. The agriculture systems had no trees. The tree density within each agroforestry alternative was assumed to stay constant throughout the simulation period while the forestry alternatives assumed standard management practice which typically included a process of thinning to reduce the tree density over time. The simulation period considered was 80 years. A full description has previously been provided (Crous-Duran et al. 2018), but a brief description of each case study is presented here for clarity.

Iberian wood pastures in Portugal and Spain

Iberian *dehesas* and *montados* occupy an area of around 3.04 million hectares (García de Jalón et al. 2018) and are characterized by low trees densities (20-50 trees ha^{-1}) combined with agriculture and/or pastoral activities (Pinto-Correia and Fonseca 2004). Two main tree species are dominant: holm oak (*Quercus rotundifolia* L.) and cork oak (*Quercus suber* L.). The dominant tree defines the main economic activity: cork *montado* where the main source of revenue is cork extraction and holm *montado* where the main activity is grazing livestock (cattle and/or Iberian pigs). In both cases cultivation, other than to reseed grassland is nowadays rare, and the main understory use remains livestock grazing (Pinto-Correia et al. 2011). The six management alternatives analysed in the *montado* case study included a pure pasture option (no-trees, MONTPT-A), four agroforestry options with 50 (MONTPT-AF1), 100 (MONTPT-AF2), 150 (MONTPT-AF3) and 200 (MONTPT-AF4) trees ha^{-1} and a higher density pure plantation (forestry alternative, MONTPT-F), with 505 trees planted followed by a thinning regime of 435, 320, 250, 200, 160 residual tree density in years 9, 45, 55, 65 and 75 respectively. It is considered that a regular light pruning occurs every 12 years removing 10% of total biomass (Olea and San Miguel-Ayanz 2006).

Grazed cherry orchards in Switzerland

Fruit orchards are of great importance in Switzerland (Sereke et al. 2015) and in Central Europe covering around 0.4 million ha of agricultural land (Eichhorn et al. 2006). These systems, typically with a tree density between 20 and 100 trees per hectare can incorporate an understory crop of grass and/or crops (Herzog 1998). The main tree species include apple trees (*Malus* spp.), pear trees (*Pyrus* spp.) and/or cherry trees (*Prunus avium* L.) that were planted for fruit and timber production. Management options considered: a pure pasture (no-trees, CTCH-A); four agroforestry alternatives with 26 (CTCH-AF1), 52 (CTCH-AF2), 78 (CTCH-AF3) and 104 (CTCH-AF4) trees ha^{-1} and a forestry/orchard alternative (CTCH-F) with an initial tree density of 690 trees ha^{-1} followed by a thinning regime of 395, 270, 190, 142 and 100 residual tree density in years 10, 15, 25, 35 and 50 respectively.

Silvo-arable systems in the UK

The silvo-arable systems analysed for the UK (SAFUK) is based on an experimental arrangement planted in 1992 as part of the UK silvo-arable network. Four poplar hybrids were planted in lines with an intercropping area allocated to a crop rotation comprising wheat (*Triticum* spp.), barley (*Hordeum vulgare* L.), oats (*Avena sativa* L.) or oilseed rape (*Brassica napus* L.). This system is not a traditional agroforestry system and, consequently, it is rarely seen in the UK (Burgess et al. 2005). For this system, the simulation period included four rotations of 20 years each with trees being replanted at the end of every rotation. In the arable option (no-tree alternative, SAFUK-A) the area is occupied 100% by the crop while in the forestry option the trees were also arranged in lines using the same spacing as in the agroforestry alternatives (10 × 6.4 m) with fallow alleys between the lines. The agroforestry tree densities analysed were 56 (SAFUK-AF1), 78 (SAFUK-AF2), 104 (SAFUK-AF3) and 156 (SAFUK-AF4) trees ha^{-1} where tree density was reduced by increasing the distance within the lines maintaining a consistent crop area as 80% of the total area. The forestry alternative (SAFUK-F) started with an initial tree density of 1250 trees ha^{-1} followed by a thinning regime of 938, 703, 352, 158 residual tree density in years 6, 8, 10 and 12 respectively.

Short rotation coppice systems in Germany

The growing of rows of short rotation coppice systems (SRC) within an arable field is an innovative system still not widely represented in Europe. The system consists in planting fast-growing trees in lines for obtaining biomass for energy between cultivated areas. However, these systems seem an interesting option for obtaining energy in rural areas occupying an area of around 6600 ha in Germany (Becker et al. 2019). The main tree species used in these systems include poplar variety Max 1 (*Populus nigra* × *Poplar maximowiczii*) and Fritzi-Pauley (*Poplar trichocarpa*) and black locust (*Robinia pseudoacacia*). Trees are planted in high densities (around 9000 trees ha^{-1}) in 11-metre-wide rows with crop alleys ranging in widths from 96 m to 24 m (96 m (SRCDE-AF1); 72 m (SRCDE-AF2); 48 m (SRCDE-AF3) and 24 m (SRCDE-AF4) in addition to the no-tree/arable (SRCDE-A) and a forestry/pure SRC (SRCDE-F) alternative.

In the crop alleys several crops are grown. For this study winter wheat (*Triticum aestivum*) and sugar beet (*Beta vulgaris*), grown alternatively, were considered. The tree coppicing rotation was assumed to be four years, enabling 20 rotations for the 80 years of the period simulated. In the simulation, in order to prevent the legal redesignation (Manning et al. 2015) of the poplar areas as “forest”, the trees were replanted every third rotation. To simulate tree-crop interaction within the SRC lines, the two double rows located in the middle on the 11 m row, were considered as pure SRC while the two double rows located on the sides were considered to interact with the crop.

Integration of regulation services in a simulation model

In a previous study, the Yield-SAFE model was used to model the provisioning services of the six systems in the four case-study sites (Crous-Duran et al. 2018). For the purposes of this paper, the model was developed to integrate three RES. These were 1) the control of soil erosion by water; 2) regulation of water quality by minimising nitrate leaching and 3) climate regulation via sequestration of carbon in the soil and as biomass (**Table 13**). The method for quantifying these regulating services are described in turn.

Table 13. Regulating ecosystem services considered for this study following the Common International Classification of Ecosystem Services (CICES) (v5.1), the indicators used, and the method implemented in Yield-SAFE model.

Section: Regulation & Maintenance						
Division	**Group**	**Class**	**Indicator**	**Method**	**Unit**	**Reference**
Regulation of physical, chemical and biological conditions	Regulation of baseline flows and extreme events	Control of erosion rates	Soil Erosion	RUSLE equation	Mg soil ha^{-1} yr^{-1}	Panagos et al. 2015e
	Water conditions	Regulation of physical, chemical, biological conditions	Nitrate leaching	Nitrogen balance	Kg N ha^{-1} yr^{-1}	Palma et al. 2007a
	Atmospheric composition and conditions	Regulation of chemical composition of atmosphere and oceans	Carbon sequestered	Yield-SAFE	Mg C ha^{-1} yr^{-1}	Palma et al. 2017b

Regulation of soil erosion

The inclusion of the regulation of soil erosion within Yield-SAFE was based on the revised universal soil loss equation (RUSLE). The equation estimates long-term average annual soil loss by sheet and rill erosion and has been the most frequent model used for this purpose (Panagos et al. 2015e). The RUSLE equation (**Equation 3**) was implemented into Yield-SAFE model and calculated following the approach used in (Palma et al. 2007a) with the exception of the cover management factor (C factor) that for the present study varied depending on the type and age of vegetation and on the disposition of the trees related to the crop.

$$A = R\ K\ L\ S\ C\ P \qquad \text{Equation 3}$$

Where A is the estimated average soil loss due to water (Mg soil ha^{-1}); R the rainfall erosivity factor calculated over one year (in MJ mm ha^{-1} day^{-1}), K the soil erodibility factor (in Mg h MJ^{-1}mm^{-1}), LS is the slope-length factor (unitless); C the cover management factor (unitless), and P the erosion control practice factor (unitless).

The erosivity factor (*R*) is defined as the sum of the average of the product of the kinetic energy of precipitation (erosive events) by the maximum intensity for a period of 30 minutes (I_{30}). The values of *R* for the four study sites were extracted from adding the monthly values reported in (Ballabio et al. 2017). The soil erodibility factor (*K*) is related to the susceptibility of soil to erode and is linked to physical properties such as the organic matter content, soil texture, soil structure and permeability. The values for *K* for the four study sites were derived from a soil erodibility map for the European Union (Panagos et al. 2014). The combined LS-factor describes the effect of slope and length on soil erosion, and the value for the Portuguese, English and German study sites were extracted from the European LS-factor map (Panagos et al. 2015b) and from Kay *et al.* (Kay et al. 2018b) for the Swiss case study. The value of *C* depends on land use and was dependent on the land use scenario within each location. Depending on the type and age of the vegetation, reference values were taken from Panagos *et al.* (Panagos et al. 2015c). A value corresponding to "forest" was considered for the C_{tree} factor (C_{tree} = 0.03) (**Table 14**).

Table 14. Values used for the RUSLE equation factors for the four agroforestry systems assessed.

Factor	**MONTPT**	**CTCH**	**SAFUK**	**SRCDE**	**Units**	**Reference**
R	519	500	265	426	MJ mm ha^{-1} $year^{-1}$	Ballabio et al. 2017)
K	0,0305	0,055	0,0305	0,024	t h $MJ^{-1}mm^{-1}$	Panagos et al. 2014, Kay et al. 2018b
LS	0,3	0,3	0,3	0,3	unitless	Panagos et al. 2015b, Kay et al. 2018b
C_{tree} **C_{crop}** **C_{fallow}**	Cork oak: 0.03 Iberian pastures: 0.15 Fallow: 0.15	Cherry tree: 0.03 Swiss pastures: 0.15 Fallow: 0.15	*Populus* spp*:* 0.03 Wheat: 0.21 Barley: 0.21 Oilseed: 0.28 Grass: 0.15	*Populus* spp: 0.03 Wheat: 0.21 Sugar beet: 0.34 Grass: 0.15	unitless	Panagos et al. 2015c
P	1	1	1	1	unitless	Panagos et al. 2015d

For the agroforestry alternatives the C-factor was calculated considering the C-factors associated with the crop (C_{crop}) and the trees (C_{tree}) following Palma *et al.* (2007a). While C_{crop} remained constant for a specific crop, C_{tree} varied depending on the growth of the trees. For the Portuguese and Swiss traditional systems, the yearly change in the area of canopy cover was used. The tree areas in the English silvo-arable and the German short rotation coppice systems were pre-determined as the trees are arranged in lines and the potential canopy cover is limited. While not covered by canopies the remaining tree area received a C-factor value as if it was covered by fallow (C_{crop} = 0.15). The final C-factor of the system was estimated combining the areas occupied by tree and/or crop following **Equation 4:**

$$C = (C_{crop} * Crop\ area) + (C_{tree} * Canopy\ area) + (C_{fallow\ (or\ pastures)} * (Tree\ area - Canopy\ area)) \qquad \text{Equation 4}$$

The P-factor in the RUSLE equations relates to erosion control practices and was assumed to equal 1 in each system as no additional erosion control practices were reported.

Regulation of nitrate leaching

The approach suggested by Palma *et al.* (2007a) for assessing the nitrate leached was followed and implemented into the Yield-SAFE model. In this approach, the estimated nitrate leached annually (kg N ha^{-1} yr^{-1}) depended on the nitrogen balance, the water flow to groundwater and the soil water content at field capacity. The potential inputs to the nitrogen balance were fertilization, atmospheric deposition, biotic fixation and mineralization. The potential nitrogen outputs were the processes of denitrification, volatilization, crop and tree uptake and immobilization. The quantity of nitrate leached was estimated as shown in **Equation 5**:

$$N_{leach} = 4.43\ N_{bal}\ EF \qquad \text{Equation 5}$$

where N_{leach} is the nitrogen leached (kg N ha^{-1} yr^{-1}); N_{bal} is the nitrogen balance (kg N ha^{-1} yr^{-1}) and EF is the annual soil water exchange factor (unitless). The value of EF depended on the calculated annual flow to groundwater (F_{gw}; in mm) and the soil water content at field capacity (*FC*; in mm) and is determined by the Yield-SAFE as shown in **Equation 6 and Equation 7**:

$$\text{If } F_{gw} / FC \geq 1, \text{ then } EF = 1 \qquad \text{Equation 6}$$

$$\text{If } F_{gw} / FC < 1, \text{ then } EF = F_{gw} / FC \qquad \text{Equation 7}$$

Annual values for groundwater recharge were calculated as the sum of daily estimates derived from the Yield-SAFE model. The value of the nitrogen balance (N_{bal}) was determined as shown in **Equation 8**:

$$N_{bal} = (N_{fert} + A_{dep} + N_{fix} + N_{min}) - (D + V + U + I) \qquad \text{Equation 8}$$

where N_{fert} is the addition of nitrogen fertilizer (mineral and organic), A_{dep} is the atmospheric deposition, N_{fix} is the biotic nitrogen fixation, N_{min} is the mineralization, *D* is the denitrification, *V* is the volatilization, *U* is the crop/tree nitrogen uptake and *I* is the nitrogen immobilization (all units are in kg N $ha^{-1} yr^{-1}$). In long term assessments of regular cropping rotations, the nitrogen from mineralization (N_{min}) and immobilization (I) are considered to be in equilibrium and therefore are not considered in the nitrogen balance (Noy-Meir and Harpaz 1977; Vlek et al. 1981). Although the inclusion of trees on arable land could modify this equilibrium through the additional nitrogen supplied by leaf fall or root mortality, these effects were not included in the Yield-SAFE model.

Nitrogen atmospheric deposition (A_{dep}), was obtained by summing values of oxidized and reduced nitrogen deposition from the European Monitoring and Evaluation Program (EMEP 2018). N_{fix} was estimated to be 1 kg N $ha^{-1} yr^{-1}$ (Palma et al. 2007a) and an average value for denitrification (*D*) of 30 kg N ha^{-1} yr^{-1} was considered for all the sites (Palma et al. 2007a). Volatilization (*V*) was considered to be 5% of nitrogen fertilization (N_{fert}) (Van Keulen et al. 2000). The nitrogen content of crops and trees and the amount of fertilizer per crop applied is shown in **Table 15**. It was considered that no extra fertilization was applied during the afforestation process and that there was no additional nitrogen input from the urine and faeces of livestock.

Table 15. Nitrogen content for the crops and trees and nitrogen fertilization considered in this study.

Crop (units)	**N content aboveground (0-1)**	**N content Roots (0-1)**	**N fertilization applied (kg N ha^{-1} yr^{-1})**	**Reference**
Montado grass	0.02	0.02	0	Otieno et al. 2011
Swiss grass	0.03	0.03	0	Büchi et al. 2015
Wheat UK	0.022	0.006	175	García de Jalón et al. 2017b
Barley UK	0.01	0.004	145	
Oilseed UK	0.018	0.007	200	
Sugar beet DE	0.00265	0.00265	120	Draycott 2006
Holm oak	0.012	0.0158	0	Bazot et al. 2013
Cherry tree	0.012	0.0045	0	Morhart et al. 2016
Poplar spp.	0.0099	0.004	0	Euring et al. 2016

Carbon sequestration

The capacity of the soil and above and belowground biomass to store carbon was derived using a soil carbon model (RothC, Coleman & Jenkinson 2014) that simulates soil organic changes, integrated by Palma et al. (2017b) in the Yield-SAFE model. The integration included estimates of tree and crop inputs into soil including leaf fall and root mortality. Long-life products, such as timber, were considered to store carbon, whilst the rapid turnover of short-life products such as grain, cherries, sugar beet, or meat (estimated through grass growth) meant that they were not included. For the silvo-pastoral systems (*montado* and Swiss orchards) the excrements of the potential livestock grazing were considered to be organic input material for the soil model. A carrying capacity of 0.5 and 1 livestock units (LU) were considered for the Portuguese and Swiss silvo-pastoral systems respectively, whereas carbon in excrements was estimated as 0.99 kg C LU^{-1} day^{-1} considering a yearly undiluted excreta volume of 19.35 m^{3} year^{-1} (Steven et al. 2009), a bulk density of 0.4 kg m^{-3} (Agnew et al. 2003) and a carbon concentration of 0.047 kg C kg excrement^{-1} (Van Horn et al. 1994).

Use of the model

Once the model was developed, the next step was to collect the weather, soil, crop, tree and livestock input data needed for the Yield-SAFE model for each site. Weather data (daily solar radiation, minimum and maximum temperature, rainfall, relative humidity and wind speed) were extracted from the Climate Picker tool – CliPick (Palma 2017a). The soil texture and depth were derived from field data from the case study sites and management practices such as the frequency and intensity of thinning or pruning were determined from local experts.

The next step was to calibrate the model so that the yields of the agricultural and forest system matched measured values, as described in Crous-Duran et al. (2018). The total period of simulation was 80 years. The last step was to run the model for the different tree densities described in a previous section of this chapter.

Results

Soil erosion

At each case study site, the lowest rates of soil erosion were associated with the greatest tree cover (**Figure 12**). The greatest soil erosion was predicted for the Swiss site, ranging, over 80 years, from 98 Mg soil ha^{-1} for the pure-pasture alternative (CTCH-A) to 83 Mg soil ha^{-1} for the tree-only system (CTCH-F). The Swiss agroforestry systems had intermediate values of 93.5, 90.3, 88 and 85 Mg soil ha^{-1} corresponding to an increasing in tree density, equivalent to soil erosion reductions of 4%, 7%, 10% and 13% compared to the pure pasture alternative (CTCH-A).

The next highest values of soil erosion occurred in the *montado* system with soil losses, for the 80-year period of simulation ranging from 53 Mg soil ha^{-1} for pasture system (MONTPT-A) to 33 Mg soil ha^{-1} for the forest system (MONTPT-F). The presence of trees in intermediate densities reduced soil erosion by 7, 14, 20 and 26% in the 50 (MONTPT-AF1), 100 (MONTPT-AF2), 150 (MONTPT-AF3), and 200 trees ha^{-1} (MONTPT-AF4) alternatives respectively.

In the UK, the predicted soil erosion for the period simulated ranged from 41 Mg ha^{-1} in the arable system (SAFUK-A) to 13 Mg ha^{-1} of the forestry alternative (SAFUK-F). Again, the agroforestry options led to intermediate values of 30 Mg ha^{-1} in the 39 trees ha^{-1} agroforestry (SAFUK-AF1), 24 Mg ha^{-1} in the 78 trees ha^{-1} agroforestry (SAFUK-AF2), 21 Mg ha^{-1} in the 104 trees ha^{-1} agroforestry (SAFUK-AF3) and 19 Mg ha^{-1} in the 154 trees ha^{-1} agroforestry alternative (SAFUK-AF4). Compared to the arable alternative, these values represented a reduction of 68%, 27%, 43%, 50% and 55% respectively.

Finally, in Germany, the pure SRC alternative in Germany (SRCDE-F) showed a reduction of soil erosion of 82% (from 43.86 to 8.25 Mg soil ha^{-1}), compared to a pure arable alternative (SRCDE-A), the agroforestry SRC lines reduced 17%, 22%, 32% and 51.8% if the distance between lines is 96 m (SRCDE-AF1), 72 m (SRCDE-AF2), 48 m (SRCDE-AF3) and 24 m (SRCDE-AF4) respectively.

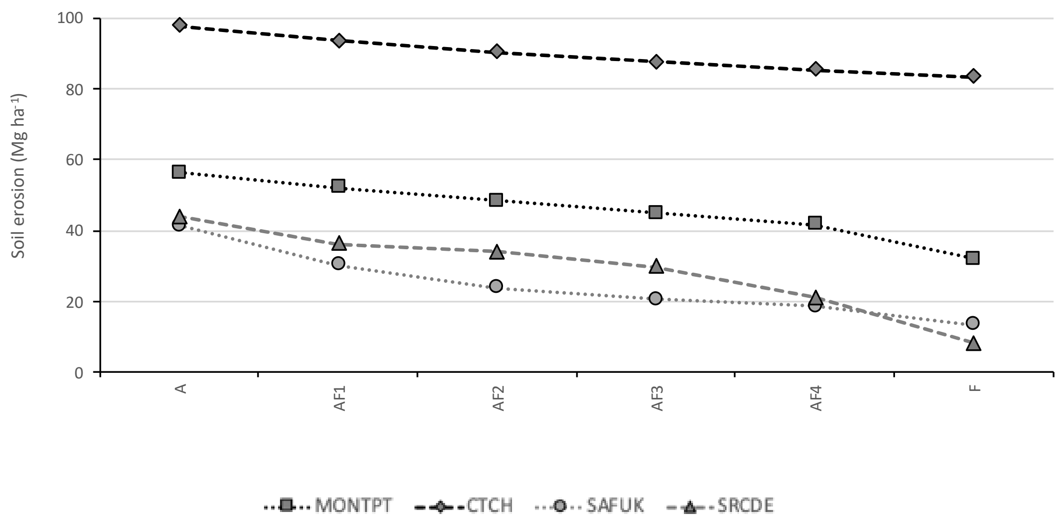

Figure 12. Accumulated soil loss for the period of 80 years (in Mg soil ha-1) for six different management alternatives in increasing tree densities across Europe. For the systems in Portugal (MONTPT), Switzerland (CTCH), the UK (SAFUK) and Germany (SRCDE). MONTPT: 0 trees ha-1 (A); 50 trees ha-1 (AF1); 100 trees ha-1 (AF2); 150 trees ha-1 (AF3); 200 trees ha-1 (AF4) and forestry (F). CTCH: 0 trees ha-1 (A); 26 trees ha-1 (AF1); 52 trees ha-1 (AF2); 78 trees ha-1 (AF3); 104 trees ha-1 (AF4) and forestry (F). SAFUK: 0 trees ha-1 (A); 56 trees ha-1 (AF1); 78 trees ha-1 (AF2); 118 trees ha-1 (AF3); 156 trees ha-1 (AF4) and forestry (F). SRCDE: pure arable (A); alley widths of 96 m (AF1); 72 m (AF2); 48 m (AF3); 24 m (AF4) and Pure SRC (F).

Nitrate leaching

Among the six management options the greatest rates of nitrate leaching were predicted for arable systems in the German and the English case studies (**Figure 13**). Because no fertilizer was applied in the Portuguese *montado* and the cherry orchards systems, the rates of leaching were minimal in those systems (Palma et al. 2007c).

For the silvo-arable system in the UK, the presence of trees supposed a reduction of nitrate leaching around cumulated after 80 years of 75%: from the 2168 kg N ha^{-1} presented in the arable alternative (SAFUK-A) to the 552, 551, 520 and 487, kg N ha^{-1} for the agroforestry alternatives (SAFUK-AF1, AF2, AF3 and AF4) respectively. Also, for this system, the nitrate leached in the forestry alternative (SAFUK-F) was also reduced to a 98% (36 kg N ha^{-1}) compared to the arable alternative.

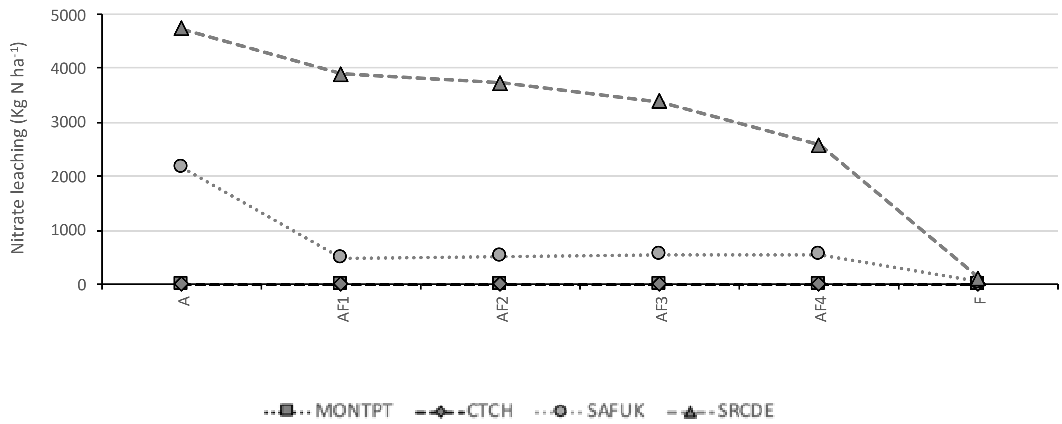

Figure 13. Accumulated nitrate leached for the period of 80 years (in kg N ha^{-1}) for six different management alternatives in increasing tree densities across Europe. For the systems in Portugal (MONTPT), Switzerland (CTCH), the UK (SAFUK) and Germany (SRCDE). MONTPT: 0 trees ha^{-1} (A); 50 trees ha^{-1} (AF1); 100 trees ha^{-1} (AF2); 150 trees ha^{-1} (AF3); 200 trees ha^{-1} (AF4) and forestry (F). CTCH: 0 trees ha^{-1} (A); 26 trees ha^{-1} (AF1); 52 trees ha^{-1} (AF2); 78 trees ha^{-1} (AF3); 104 trees ha^{-1} (AF4) and forestry (F). SAFUK: 0 trees ha^{-1} (A); 56 trees ha^{-1} (AF1); 78 trees ha^{-1} (AF2); 118 trees ha^{-1} (AF3); 156 trees ha^{-1} (AF4) and forestry (F). SRCDE: pure arable (A); alley widths of 96 m (AF1); 72 m (AF2); 48 m (AF3); 24 m (AF4) and Pure SRC (F).

For the German system, nitrate leaching levels in the arable alternative (SRCDE-A) were higher (4726 kg N ha^{-1}) compared to the agroforestry alternatives (SRCDE-AF1, AF2, AF4 and AF4) that ranged between 3885 and 2574 kg N ha^{-1} corresponding to a reduction in nitrate leaching of 18, 21, 29, 46% if compared to arable. In the forestry alternative

(SRCDE-F) the nitrate leached was nearly null (reduction of 98%) presenting an accumulated value of 113 kg N ha^{-1} for the 80 years simulated.

Carbon sequestration

The greatest level of carbon sequestration, aggregated over 80 years, were predicted for the poplar systems in the UK (**Figure 14**). The predicted carbon storage in the agricultural systems remained relatively constant, ranging from an increase of 0.1% in Portugal, an increase of 2.7% in Switzerland and a reduction of 5.5% in the UK, and 6.0% in Germany. Across the agroforestry systems, an increase in the density of trees led to an increase in carbon storage. At the Portuguese, Swiss, and British sites, the agroforestry option with the highest density of trees (MONTPT-AF4, CTCH-AF4, SAFUK-AF4) had higher predicted levels of carbon sequestration than the forestry option. By contrast, the forestry system in Germany (SRCDE-F) was able to sequester the triple compared to the most-dense agroforestry option (SRCDE-AF4).

Considering each case study, for the 80 years of assessment of the *montado* system, the agroforestry and forestry alternatives ranged values of carbon sequestration in total of between 24 Mg C ha^{-1} (MONTPT-AF1) and 57 Mg C ha^{-1} (MONTPT-AF4). For the cherry orchards, the values ranged from 112 Mg C ha^{-1} (for the CTCH-AF1 alternative) to 267 Mg ha^{-1} found (for the CTCH-AF4 and CTCH-F alternatives). In the English silvo-arable system, the presence of trees allows to sequester a positive amount up to nearly 400 Mg C ha^{-1} in the most tree-densed agroforestry alternative (SAFUK-AF4) and a minimum of 240 Mg C ha^{-1} in the less dense (SAFUK-AF1, 56 trees ha^{-1}). These systems have a net positive carbon sequestration compared to the “no trees” alternative that shows a loss of around 1 Mg C ha^{-1} (from an initial soil carbon of 21.5 Mg C ha^{-1} to a final carbon sequestration of 20.5 Mg C ha^{-1}). Related to the German site, the only the presence of trees ensures a positive net carbon sequestration with a minimum of 46 Mg C ha^{-1} in the less dense agroforestry alternative (96m wide lines, SRCDE-AF1) up to 122 Mg C ha^{-1} for the most-desned agroforestry alternative (SRCDE-AF4) with the pure SRC alternative (SRCDE-F) tripling this value (373 Mg C ha^{-1}).

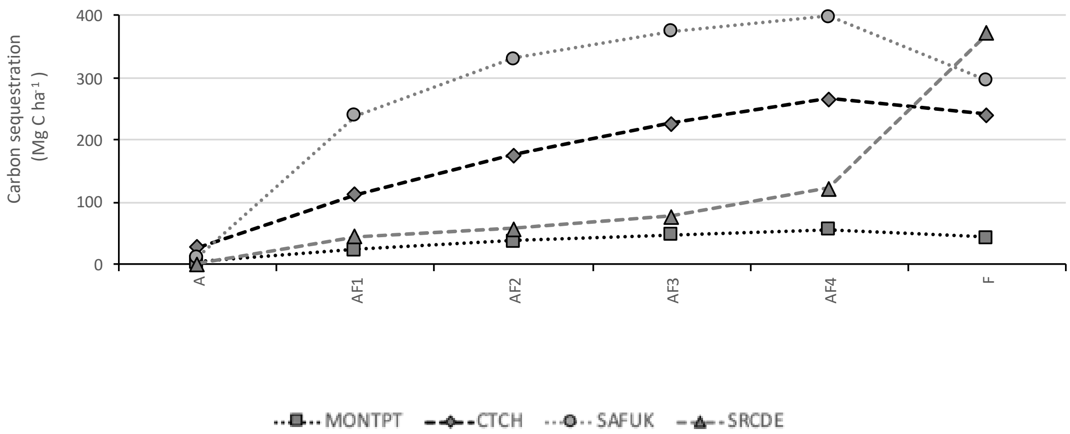

Figure 14. Carbon sequestered for the period of 80 years (in Mg C ha^{-1}) for six different management alternatives in increasing tree densities across Europe. For the systems in Portugal (MONTPT), Switzerland (CTCH), the UK (SAFUK) and Germany (SRCDE). MONTPT: 0 trees ha^{-1} (A); 50 trees ha^{-1} (AF1); 100 trees ha^{-1} (AF2); 150 trees ha^{-1} (AF3); 200 trees ha^{-1} (AF4) and forestry (F). CTCH: 0 trees ha^{-1} (A); 26 trees ha^{-1} (AF1); 52 trees ha^{-1} (AF2); 78 trees ha^{-1} (AF3); 104 trees ha^{-1} (AF4) and forestry (F). SAFUK: 0 trees ha^{-1} (A); 56 trees ha^{-1} (AF1); 78 trees ha^{-1} (AF2); 118 trees ha^{-1} (AF3); 156 trees ha^{-1} (AF4) and forestry (F). SRCDE: pure arable (A); alley widths of 96 m (AF1); 72 m (AF2); 48 m (AF3); 24 m (AF4) and Pure SRC (F).

Discussion

The results are discussed in terms of the effect of tree density on soil erosion, nitrate leaching and carbon sequestration.

Soil erosion

The lowest predicted levels of soil erosion, over 80 years, ranged from 85 Mg ha^{-1} at the Swiss site and 33 Mg ha^{-1} in Portugal, to 13 Mg ha^{-1} and 8 Mg ha^{-1} at the British and German sites respectively. The low rates at the British and German sites were partly the result of low rain erosivity (*R*) as rainfall tends to regularly be distributed through the year, at the Portuguese and the Swiss sites, much of the rainfall is concentrated into just a few months. The particularly high values at the Swiss site were a result of high soil erobility (*K*) and high slope-length (*LS*) (García-Ruiz et al. 2015).

Across the four case studies, a greater presence of trees, modelled through the effect of cover management (*C*-factor), led to a predicted reduction in the rate of soil erosion (**Figure 12**). The value of C-factor is defined as how the tree and crop elements are able to mitigate soil loss compared to bare fallow areas. Because the *C* value of 0.15 for natural grasslands in Portugal or Switzerland is close to the value for tree cover (0.03), the effect of trees on avoiding soil erosion is minimised. By contrast, the high *C* values assumed for cereals (*C* = 0.21) and sugar beet (C = 0.34) in the UK or Germany increases the importance of trees in reducing soil losses.

Expressed as an annual value, the soil erosion rates in Switzerland ranged from 1.03 to 1.20 Mg soil ha^{-1}. These rates are lower than a previously cited value of 1.8 to 2.5 Mg soil ha^{-1} yr^{-1} (Kay et al. 2018a). The difference is probably due to the high difference slopes presented in both studies and not necessarily due to a tree effect on erosion. The predicted annual soil erosion rates for the *montado* ranged from 0.41 to 0.66 Mg ha^{-1} similar to values of 0.5 and 1 Mg soil ha^{-1} reported by (Pimentel and Krummel 1987; Guerra et al. 2014). For the silvo-arable systems in the UK, the annual results of 0.15 to 0.51 Mg ha^{-1} seem to be consistent, even if slightly lower, compared to value of 0.19 to 0.38 Mg ha^{-1} reported by (García de Jalón et al. 2017b). Finally, in Germany, the annual soil erosion values estimated ranged from 0.85 to 0.20 Mg ha^{-1} soil yr^{-1}. On moderate slopes (≤5%) under SRC systems values of erosion rates expected would be on average of around 2 Mg ha^{-1} soil yr^{-1} but the rate depends on site preparation and harvesting technology (Pimentel and Krummel 1987). However, the lower values obtained in the present study could be explained by: 1) the presence of crops in between lines that has been stated can help reducing soil erosion (Tolbert et al. 2000) and 2) the reduction on the soil erodibility factor (K-factor) associated to tree barrier effect that reduces evapotranspiration and increases carbon storage (Kort et al. 1998) and that was not considered in this study.

Nitrate leaching

The predicted rates of nitrate leaching were driven by 1) the surplus in balance between the nitrogen inputs and outputs and 2) soil water surplus acting as the movement vector for nitrate percolation. Palma et al. (2007c) and Kay *et al.* (2018a) suggested that nitrate leaching is low or negligible in Mediterranean areas as precipitation rarely exceeds evapotranspiration so there is a low flow of water to groundwater. In addition, in this study the lack of any artificial nitrogen application on natural grassland in the Portuguese case study, meant that no nitrate leaching was predicted. Similarly, although the precipitation was high at the Swiss site, the lack of fertilization of the grasslands or cherry trees meant again that nitrate leaching was negligible (**Figure 13**).

By contrast in the silvo-arable site in the UK, after 80 years, the predicted rate of nitrogen leaching in the agroforestry alternatives was only 22-25% of that predicted for the arable system. This was partly due to a reduced rate of nitrogen application and the deep roots of the trees taking up some of the surplus nitrogen. Expressed as an annual value, the leaching loss in the agroforestry system of around 27 kg N ha^{-1} is similar to the value of 25 kg N ha^{-1} reported by García de Jalón et al. (2017b).

An existing study (Hartmann & Lamersdorf 2015) have demonstrated that poplars can reduce nitrate leaching to the groundwater of agricultural landscapes. In the German case study, the predicted effect of the trees was to reduce the amount of nitrate leached but this effect was not linear as low densities are proportionally able to reduce more nitrate leaching. This effect therefore suggests that initial trees have a major reduction impact of nitrogen applied as fertilizer for growing wheat. However, no nitrate leached was appreciated in sugar beet cultivation years as it has been seen that due to the long growing season and deep root system sugar beet nitrogen uptake usually exceeds fertilizer application (Sylvester-Bradley and Shepherd 1997).

The predicted reduction in nitrate leaching due to the presence of trees is in accordance with previous studies. For example, Nair *et al.* (2007) found a higher overall nutrient uptake in pastures with trees and therefore a lower nutrient concentration in soils. Hartmann *et al.* (2015) reported that poplar short rotation coppice systems were more efficient in terms of nitrogen uptake when in an agroforestry system. However, the methodology used, consisting of an annual nitrogen balance between nitrogen inputs and outputs is limited and is not able to reflect the seasonal peaks of nitrate leaching commonly associated with months with less vegetative growth or more intense rainfall events (Kay et al. 2018a).

Carbon sequestration

The predicted rates of carbon sequestration were derived from changes in the above- and below-ground biomass and the soil carbon content, which in turn depended on the inputs provided through root mortality and leaf fall (Palma et al. 2017b). Hence the maximum increase in carbon sequestration at each site is constrained by the biomass production. Across the four sites, the presence of additional trees increased total carbon sequestration until a point where additional trees are unable to capture more solar radiation and water (Crous-Duran et al. 2018).

For the simulation period of 80 years, the predicted carbon sequestration by the *montado* agroforestry systems at the dry Portuguese site ranged from 24 Mg C ha^{-1} (50 trees ha^{-1}) to 57 Mg C ha^{-1} (200 trees ha^{-1}). This is consistent with previous studies, for example, Palma *et al.* (2014) found values of around 40 Mg C ha^{-1} for 50 trees ha^{-1}. In the Swiss case study, the agroforestry system with 104 trees ha^{-1} was predicted to sequester 267 Mg C ha^{-1}, similar to the value of 260 Mg C ha^{-1} (80 years multiplied by 3.25 Mg C ha^{-1} yr^{-1}) reported by Kay *et al.* (Kay et al. 2018b). For the English silvo-arable alternative of 156 trees ha^{-1} (SAFUK-AF4) after 20 years a tree timber growth of around 0.8 m^3 tree^{-1} was expected (Crous-Duran et al. 2018). Considering a wood density for poplar of 410 kg m^{-3}, a carbon content of 49,8% of the biomass (Mathews 1993) and a tree density of 156 trees ha^{-1}, a value around 25 Mg C ha^{-1} can be estimated. Considering just timber, the results are similar to those reported by Palma *et al.* (2007c), of between 10 and 48 Mg C ha^{-1}, for a similar silvo-arable system in the Netherlands when adapted to a 20-year rotation, and by García de Jalón *et al.* (2017b) that estimated a carbon content for timber of around 22 Mg C ha^{-1}. However, in this last value, and accounting for 30% of biomass as branches and leaves, 40% of total aboveground biomass as roots and 20 Mg C ha^{-1} of initial soil carbon the total amount adds up to 68 Mg C ha^{-1} accumulated per rotation (20 years).

Crous-Duran *et al.* (2018) calibrated the Yield-SAFE model for the German short rotation coppice system. With a final tree density of 7157 trees ha^{-1}, for a 4-year rotation, the average value was of 4.9 kg tree^{-1}, corresponding to a total stand biomass of 33 Mg ha^{-1}. A similar system with approximate yields is reported by Mirck *et al.* (2016) and Kanzler and Mirck (2017), although with a stand density starting at 8,497 trees ha^{-1} and a final density of 6,295 tree ha^{-1}. These authors reported an average tree biomass of 3.59 kg tree^{-1} and a total stand biomass yield of 22.6 Mg ha^{-1}. Considering a carbon content in poplar of 50% of the total biomass (Mathews 1993), our results show levels of carbon sequestered in aboveground biomass averaging 16.5 Mg C ha^{-1}. On the other hand, for this system soil carbon levels remained nearly constant in all the alternatives analyzed in accordance with was previously found of non-significance in soil organic carbon in SRC plantations compared to adjacent crop fields (Medinski et al. 2014).

Relative effect of additional trees

Within the Yield-SAFE model, the predicted soil erosion, nitrate leaching and carbon sequestration within agroforestry systems are depend on the growth of the tree and crop elements. In general, the higher canopy and deeper rooting of trees, relative to grass and arable crops, meant that biomass production within the system increased as tree density increased.

At each site, the lowest rate of soil erosion was observed in the system with the highest tree density. However, at the Portuguese, Swiss, and English sites (where the densest agroforestry system had the same tree density as the tree-only system), the tree-only system had a lower predicted rate of soil erosion. The proportional decrease in soil erosion at each site is shown in **Figure 15**.

Comparing the tree-present alternatives to the arable alternative and considering the average decrease in soil erosion avoided, results show that the initial trees are able to avoid more soil erosion compared to the additional ones (**Figure 15A**). In this sense, the increase in tree density from A to AF1 represents an average decrease of 14% of the soil erosion while the following increases in tree density (AF2, AF3 and AF4) just represent a decrease of 7.7, 6.5 and 8.1% of soil eroded respectively.

Related to the nitrate leaching ones (**Figure 15B**), as the Portuguese *montado* and the Swiss cherry orchards do not present fertilization of the grasslands, nitrate leaching was considered negligible for these systems. For the English silvo-arable and the German short rotation case studies, the forestry alternatives (SAFUK-F and SRCDE-F) are able to nearly eliminate any trace of nitrate leaching (99 and 98% respectively). However, there is a difference between both systems related to the effect of the implementation of the first trees. In this sense, the AF1 alternative for the English case (SAFUK-AF1) is able to reduce 78% of the nitrate leaching while in the German case the reduction is only of 18% reaching the 46% for the most-dense agroforestry alternative (SRCDE-AF4). Both values are within the range previously observed of between 40 and 70% (Nair 2007; Nair et al. 2007b; Jose 2009) and up to 85% in greenhouse experiments (López-Díaz et al. 2011).

Finally, in relation to the carbon sequestration (**Figure 15C**) in all the four systems the arable alternatives are considered to be *in equilibrium* and present a null or even a slight decrease capacity to sequester carbon after 80 years of simulation. Also, the AF4 alternatives perform better than the forestry alternatives in the Portuguese *montado* (MONTPT-AF4), the Swiss orchards (CTCH-AF4) and the silvo-arable systems in the UK (SAFUK-AF4). The forestry alternative has higher performance in the German short rotation site. But again, for the four systems, an “initial tree effect” is detected meaning that the initial trees have a higher impact in terms of carbon sequestered in biomass and soil compared to the following ones as increases on average 92 Mg C ha^{-1}. This effect is very important for the silvo-arable system in the UK where the less tree-dense alternative (SAFUK-AF1) is able to sequester additional 227 Mg C ha^{-1} compared to the arable alternative (SAFUK-A).

Link to Provisioning services

Crous-Duran *et al.* (2018) showed that an increase in the tree density was associated to an increase in the amount of energy accumulated by the different alternatives and quantified as Provisioning Ecosystem Services (PES, food, materials and energy) but also that the accumulated energy per tree was reduced as tree density increased. For all the alternatives, the energy accumulated per tree was greatest in the lowest tree densities alternative (AF1, 50 trees ha^{-1} for the Portuguese *montado*; 26 trees ha^{-1} for the Swiss cherry orchards; 56 trees ha^{-1} for the English silvo-arable systems and 96 m alley widths for the German SRC system). The results were explained by the concept that the combined presence of tree and crop at the same parcel help to better use the natural resources available (soil, water and light) by both elements.

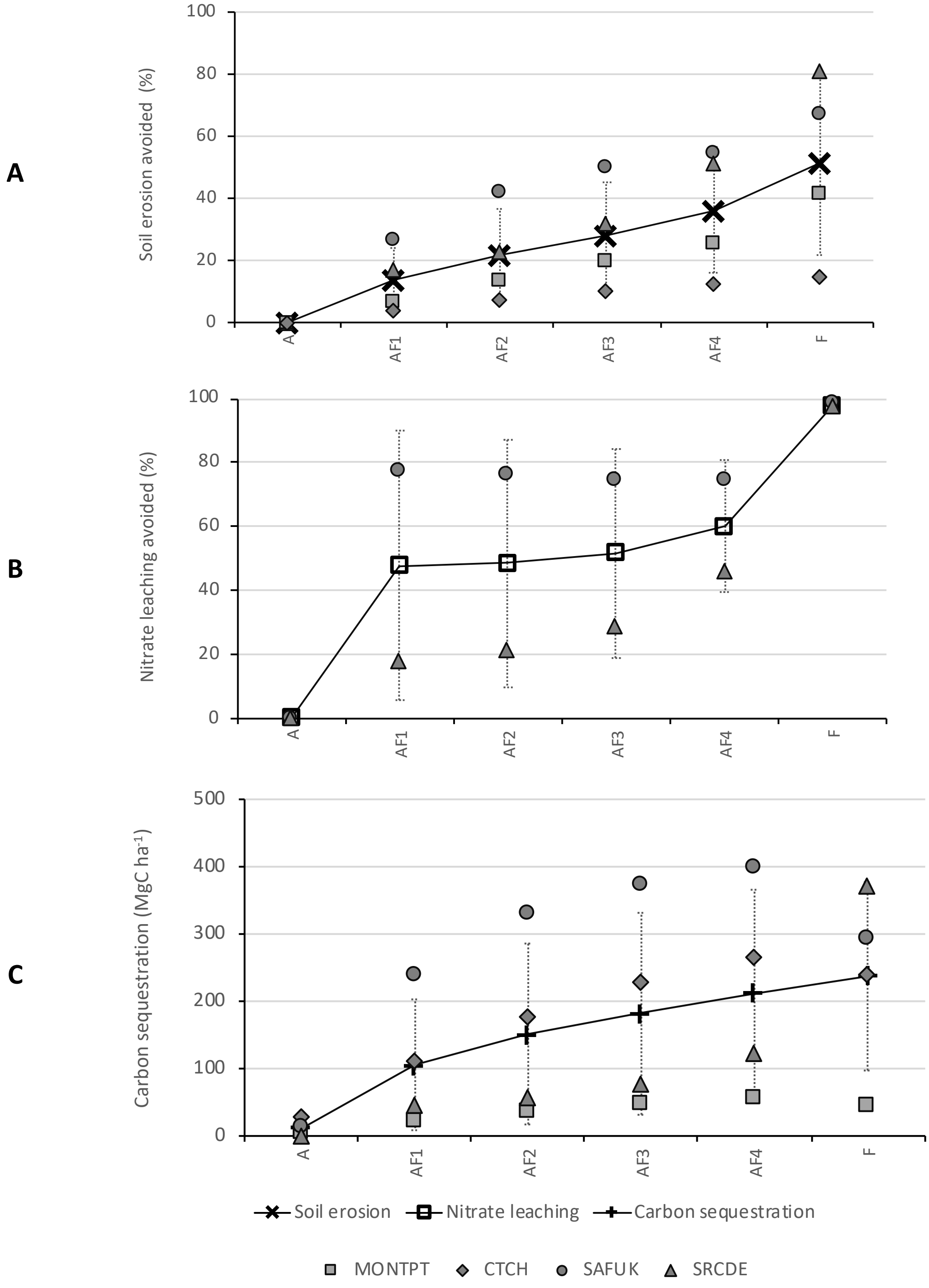

Figure 15. Average percentage of reduction of soil loss (A); Average percentage of reduction of nitrate leaching (B) and average percentage of increase in carbon sequestration (C) for six different management alternatives in increasing tree densities across Europe: for the *montado* systems in Portugal (MONTPT); the cherry orchards in Switzerland (CTCH); the silvo-arable systems in the UK (SAFUK) and the short rotation coppice in Germany (SRCDE). MONTPT: 0 trees ha-1 (A); 50 trees ha-1 (AF1); 100 trees ha-1 (AF2); 150 trees ha-1 (AF3); 200 trees ha-1 (AF4) and forestry (F). CTCH: 0 trees ha-1 (A); 26 trees ha-1 (AF1); 52 trees ha-1 (AF2); 78 trees ha-1 (AF3); 104 trees ha-1 (AF4) and forestry (F). SAFUK: 0 trees ha-1 (A); 56 trees ha-1 (AF1); 78 trees ha-1 (AF2); 117 trees ha-1 (AF3); 156 trees ha-1 (AF4) and forestry (F). SRCDE: 0 trees ha-1 (A); alley widths of 96 m (AF1); 72 m (AF2); 48 m (AF3); 24 m (AF4) and Pure SRC (F).

But increasing the tree density was also associated to an increase in the tree inter-specific competition that leads to a reduction of the biomass stored by each tree (Graves et al. 2007, 2010). In the present study, as the methodologies implemented are related directly to tree and crop element growth, results become consistent. In general, all the systems present higher capacity to mediate soil erosion, nitrate leaching and sequester carbon as tree density increases meaning that the combination at the same area of trees an crop increases not just the supply of food, energy and materials but also the capacity of the system to regulate the environment. But both effects, if allocated to each tree, decreases as more trees are added into the field. Therefore, the potential of trees to act as suppliers of goods and regulators of the ecosystem conditions is different depending on the total number of trees already present in the system. Tree density also plays an important role as growth rate has an effect on the amount and rate of the RES supplied, especially when there is a reduction in individual tree growth associated to an increase in tree density (Balandier et al. 2003; van der Werf et al. 2007). This growth rate is also affected by other aspects that need to be considered for a better understanding of the magnitude and direction of the beneficial effects supplied. These aspects can be related to weather and soil conditions of the locations of the systems, the tree and crop varieties selected or the distribution of both in the fields. In this sense, the holm oak can be considered as slow-growing tree species needing around 80 years before being considered mature. Cherry trees are intermediate with 40 to 60 years until tree harvest and the poplar varieties used in the English silvo-arable system and the German bio-energy system offer much shorter rotations: 20 years in the UK and to 12 years (3 rotations of 4 years each) in the German case study. For the supply of environmental benefits, the time needed the positive effects become noticeable would be different.

Another aspect to consider is that the inclusion of woody elements in farmland in agroforestry systems helps storing carbon in a) long term biomass compared to no-tree systems but also in b) the soil (Cardinael et al. 2017). The inclusion of tree elements increases storage compared to no-trees alternatives and the rate of sequestration increases as tree elements grow (Stephenson et al. 2014). However, an important point of discussion is the life-expectancy of the carbon sink present in this biomass and in the wood products associated. Carbon stored after harvest in SRC systems can be considered as very short (around 1-3 years) or short for poplar cheap wood (5-10 years); the carbon stored in wood used in furniture (as expected for the cherry tree wood) will be stored for decades or even if trees are not harvested storage can be endured for even centuries. In this study, the potential carbon sequestered by the biomass grown by the systems is estimated, but, the final destination of this biomass should be considered for a better perception of the potential capacity of these systems to have an impact on the reduction of the GHG emissions and on mitigating Climate Change.

Finally, when considering how tree density affects the overall biomass growth of a system and the related capacity to mediate the bio-physical conditions, there is a need to consider the microclimatic effects associated to the tree canopy cover presence. Currently the Yield-SAFE model considers competition between tree and crop for water and light and does not include the capacity of trees to grow deeper roots, increase height or leaf area in case of intraspecific competition. The assessment presented neither considers the potential modification of air temperature, wind speed and the consequent effects on the vapour pressure deficits or soil water due to the canopy presence that can have an impact on the productivity of the elements (Cubera et al. 2009; Rivest et al. 2011; López-Díaz et al. 2015). During the AGFORWARD project recent research included the implementation of these microclimatic effects due to tree presence into the Yield-SAFE model (Palma et al. 2016), however, the consequences on the potential supply of Provisioning or Regulating Ecosystem Services have not been tested and should be considered in future research.

Conclusions

Environmental benefits offered by four agroforestry systems in different biogeographical conditions are assessed comparing the results with no-trees and high tree density alternatives allowing to better understand the role played by trees in arable land. The results of this integrated analysis show an increase in regulation and maintenance of ecosystem services either in air, soil and water compartments related to tree density. In this sense, an increase in tree density resulted in: 1) a maintenance of the soil structure and quality as trees act as agents avoiding soil erosion due to the presence of the canopy and permanent grass cover in the tree line in case of linear systems; 2) an increase in the water quality as trees uptake the excess of nutrients needed in farming practices (recovery factor of nutrients is usually never above 0.8 (Van Keulen 1977; Van Keulen 1982; Van Keulen and Wolf 1986) and 3) a reduction of the amount of GHG in the atmosphere as the systems are able to sequester more carbon in biomass and soil. However, similarly to the assessment done of Provisioning Ecosystem Services (Crous-Duran et al. 2018) the effects are not linear. In one hand the value of lower tree densities is a strong boost in regulating services, in the other hand, lower tree densities have higher incremental value comparatively with higher tree densities.

In any case, what this study assessed is that the introduction of trees in arable land can potentially bring in addition to an increase in the supply of Provisioning Ecosystem Services show in a previous study (Crous-Duran et al. 2018) by benefiting from the better use the natural resources available (soil, water and light) between tree and crop, an improvement in the supply of Regulating Ecosystem Services i.e. the potential capacity of the system to regulate the environment and reduce soil erosion, the nitrate leached and increasing the amount of carbon sequestered, a win-win situation that, well managed, can converge a financial benefit for the land-owner and an environmental/societal benefit for the general public reinforcing the concept of agroforestry as a valuable sustainable intensification practice.

Author Contributions

Concept: J.C.D and J.H.N.P; Methodology: J.C.D, A.R.G, SGDJ and J.H.N.P; Formal analysis: J.C.D and J.H.N.P; Original draft preparation: J.C.D, J.H.N.P and S.GDJ; Writing, review and editing: J.C.D, J.H.N.P, P.J.B, S.GDJ, S.K, M.T, A.R.G and M.G.; Funding acquisition: P.J.B, M.T and J.H.N.P.

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Highlights

1- The methodology includes the implementation to the Yield-SAFE model of three methodologies for assessing soil erosion, nitrate leaching and carbon sequestration.

2. The model was applied to four different agroforestry systems presenting different tree and crop elements and representing various biogeographical regions in Europe.

3. The effect of tree density was assessed by considering 6 different alternatives ranging, from no-tree presence (arable/pure pastures, A) to a forestry alternative (F) with 4 middle tree-densities (agroforestry alternatives, AF).

4. In all the systems the presence of trees improved not only the biomass growth of the system but also the supply of regulating ecosystem services being this effect especially important for the initial trees.

5. The supply of RES is directly related to the aspects that affect biomass growth such as weather and soil conditions, tree/crop varieties or management.

Abbreviations

AFS - Agroforestry systems

CICES - Common International Classification of Ecosystem Services

CTCH - Cherry tree pastures in Switzerland

MONTPT - *montado* in Portugal (MONTPT)

SAFUK -silvo-arable systems in the United Kingdom

SRCDE- short rotation coppice systems in Germany

PES – Provisioning Ecosystem Services

RES - Regulating Ecosystem Services

RUSLE - revised universal soil loss equation

Chapter 5| Assessing food sustainable intensification potential of agroforestry using a carbon balance method

Based on **Crous-Duran J**, Graves AR, García de Jalón S, Paulo JA, Tomé M, Palma JHN. Assessing food sustainable intensification potential of agroforestry using a carbon balance method. *IForest* **12**, 85–91 (2019).

Abstract

Food security, climate change mitigation, and land use challenges are interlinked and need to be considered simultaneously. One possible solution is sustainable intensification, which is the practice of increasing food production per area of land whilst also reducing the environmental impacts associated with this. Agroforestry has been stated to be one practice that meets this definition. In this study, a new methodology is presented to assess the potential of different management options as sustainable intensification practices. The methodology is based on comparing the carbon emissions associated with the production of food and the carbon sequestered for that same activity for a particular quantity of food produced over a specific area and over a specific time. The resulting indicator, the "carbon balance" is the difference between the greenhouse gasses emitted (considered as negative values) and carbon sequestered (positive values) estimated in Mg CO_{2eq} per Mg of food produced on one hectare of land for one year. The carbon balance quantifies the global warming potential associated with sustainable intensification by integrating a process-based model with life cycle analysis and is able to estimate above- and below-ground biomass and soil carbon content. This methodology is tested in Portugal for wheat production under crop monoculture and agroforestry systems. The results show agroforestry to be a suitable practice for sustainable intensification compared to a crop monoculture as it maintained (and even slightly increased) wheat yields whilst providing a positive carbon balance from year 50 onwards of approximately 1 Mg of CO_{2eq} sequestered per Mg of wheat produced.

Keywords

Climate change mitigation; food security; land-use occupation; regulating ecosystem services; soil fertility; Life cycle analysis; Yield-SAFE; Clipick; carbon sequestration

Introduction

The land required for food production occupies 38% of the total land area of the world and agriculture is responsible for 19% to 29% of total global anthropogenic greenhouse gas (GHG) emissions (Foley 2011; Vermeulen et al. 2012). The GHG associated with agriculture is also steadily increasing due to the need to feed a growing global population (Tilman et al. 2011). In 2009 the Food and Agriculture Organization of the United Nations (FAO 2009) stated that the challenges associated with the reduction of GHG emissions, food security, and land use should be dealt with simultaneously on the same land parcel. Since then, even though the concept is unclear even for experts (Petersen and Snapp 2015), sustainable intensification (SI) has been identified as a strategy for improving food security while maintaining biodiversity and ecosystems services (Godfray et al. 2012; Godfray and Garnett 2014).

Agroforestry is one of the most common land use practice world-wide (den Herder et al. 2017). It consists of integrating woody vegetation with crop and/or animal production and it has been often referred to as an example of a SI practice that is able to satisfy food security concerns whilst producing other benefits (Glover et al. 2012; Godfray et al. 2012). It can provide higher yields of provisioning ecosystem services (food, materials, and energy) in comparison with obtaining the same provisioning ecosystem services from monoculture systems (Graves et al. 2010; Torralba et al. 2016; Crous-Duran et al. 2018). At the same time agroforestry can reduce soil erosion, nitrate leaching (Palma et al. 2007c), greenhouse gas emissions and potentially achieve net carbon sequestration per unit of product (Godfray et al. 2012; Vermeulen et al. 2012). In Portugal, agroforestry systems are extensively found in the form of the *montado*, which combines low density (less than 80 trees per hectare) spatially dispersed cork oak trees (*Quercus suber* L.) and holm oak trees (*Quercus ilex* subsp *rotundifolia* L.) with pasture on which livestock graze freely. The *montado* occupies around 0.75 million ha in Portugal and its equivalent system in Spain (*dehesa*) around 1.5 million ha. The most important product provided by the *montado* is cork, and Portugal is the world's largest cork producer with 49.6% of world cork production. The system provides other Provisioning Ecosystem Services such as wood, charcoal, crops, fodder, meat, dairy products, honey, mushrooms, and medicinal and aromatic plants (Pereira et al. 2003;

Pinto-Correia et al. 2011; Moreno et al. 2017). In the 1930s, in order to increase internal food production, cereal cultivation under the trees was promoted, and a silvo-arable version of the *montado* developed. However, this intensification process (including mechanisation and fertilisation) occurred without considering the fragility and low quality of the soils. As a consequence, tree density was reduced, the roots of the remaining trees were affected, and the soil erosion increased. All together lead to a general impoverishment of the soils and a further decrease in crop yields which in the following years caused the final abandonment of the fields (Pinto-Correia 1993).

Physiological growth models are models that simulate ecological processes in order to estimate vegetative growth. These models are useful in decision-making that relates to the management of natural resources in the context of climate change (Cuddington et al. 2013). The agroforestry model Yield-SAFE is a process-based model that simulates competition between trees and crops for water and light. It has been widely used in Europe including for silvo-arable systems of poplar and cereals in the UK, Netherlands, Spain, and France (Graves et al. 2010); cherry tree pastures in Switzerland (Sereke et al. 2015) and the Portuguese *montado* (Palma et al. 2014). Recently during the AGFORWARD project (Burgess and Rosati 2018) the model was further developed and calibrated for more tree and crop species (Palma et al. 2016, 2017c).

In this study we have developed a so called “carbon balance method” which quantifies GHG emissions using Life Cycle Assessment (LCA) for estimating the Global Warming Potential (GWP) of crop and tree production as well as the positive effects of carbon sequestration. Applied to crop production, this approach could be useful for estimating the SI potential of different land management practices. The method follows FAO which advocates reducing GHG emissions advice (for climate change mitigation), increasing food production (for food security), and reducing land occupation (for reduced land use change) simultaneously by comparing the level of greenhouse gas emitted by the practice (using the GWP impact from a LCA approach) with the carbon sequestered by the system (using a process-based growth model) for the same yield of product (Mg of food) per area of land (hectare). The carbon balance of the product (in $CO2_{eq}$ Mg of food^{-1}) provides a means of comparing the impact of food production under different management practices. The methodology is tested by comparing the carbon balance for growing 1 Mg of wheat in central Portugal using two alternative management scenarios: i) a wheat crop monoculture and ii) a *montado* agroforestry system combining wheat cultivation with low density cork oak trees (*Quercus suber* L.).

Materials & Methods

Definition of management scenario options

For this study, two different management options for wheat cultivation were compared: 1) wheat monoculture crop system and 2) *montado* agroforestry system combining wheat production with cork oak. Growth of both land use alternatives were simulated using the improved version of the Yield-SAFE model (Palma et al. 2016, 2017c). Both management options were simulated for Montemor-o-Novo (Portugal, Lon: 38.72; Lat: -8.32). In this region, the most common wheat specie produced is *Triticum aestivum* (L.). Average yields from 1990 to 2011 in Portugal were around 1.4 Mg ha^{-1} $year^{-1}$ (Almeida and Maçâs 2016). In Portugal, wheat is sown in the autumn (November) when temperatures begin to fall. Vegetative growth takes place in winter and it is harvested in early summer in the first weeks of June, before the hottest months of the year (Rosado 2009). For this study, the rotation simulated was a typical wheat-wheat-fallow rotation.

Whilst the crop monoculture land use was assumed to be planted on 100% of each hectare, the agroforestry management option assumed a crop area that covered 90% with 10% of each hectare being covered by trees. These were assumed to be planted at an initial density of 200 trees ha^{-1}, and then thinned every 10 years to reach a final density of 35 trees ha^{-1} at year 70.

Global Warming Potential

The carbon balance method for this study used the conversion factors published by the IPCC (Intergovernmental Panel on Climate Change) (IPCC 2006) and considered the GWP of different gaseous emissions through a common metric ($CO2_{eq}$) which in this study expressed the potential contribution of gaseous emissions to global warming over a time horizon of 100 years (GWP_{100}).

Sources of gaseous emissions were associated with the management practices for wheat and cork oak (Kramer et al. 1999; Gonzalez-Garcia et al. 2013) including the operations for the establishment of the stand, maintenance and growth, harvesting, and the transport of system products and workforce. In agroforestry, the emissions from the cork oak stand management and wheat were considered proportionally to the area occupied by the tree and the crop components for field operations (10% and 90% respectively) but the same distance was used for the transport of system products and the workforce (5 km).

The three main GHG gases included in the study were carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). These were assessed for four main sources including: 1) from the combustion of fuels used in the field operations; 2) from the emissions related to the production and application of fertilisers and seeds; 3) from the transport from field to farm of the products and 4) from soil microbial activity.

For the wheat systems, field operations included ploughing with disk harrows at the beginning of November and harrowing at the same time as sowing two weeks later. For sowing, a fine grain seeder was used at a seeding rate of 180 kg ha^{-1} and Diammonium phosphate (18:46:0) applied at a rate of 250 kg ha^{-1}. In March, fertiliser was applied again using 170 kg ha^{-1} of urea-based fertiliser (46% N). Weeding was accomplished by using a single application of 0.4 kg ha^{-1} of herbicide 2,4-D. A combine harvester was used for harvesting the wheat (**Table 16**).

In the cork oak agroforestry systems, the time required for field operations and the rate of fuel consumption for field operations was established using national data (CAOF 2010). Field operations included soil preparation to clear vegetation, and the ripping and ploughing of all the area. It was assumed that the ploughing for the wheat and for the trees would be done at the same time.

Table 16. Source of greenhouse gas emissions from wheat production

Input production	**Quantity**	**GWP 100 (kg CO_{2eq} kg^{-1})**	**GHG (kg CO_{2eq})**
Seeds (kg ha^{-1})	180[a]	0.241[b]	43.4
Fertiliser (kg N ha^{-1})	45	2.66 [b]	119,7
Urea (kg ha^{-1}) (46%N)	170[a]	0.730[c]	124.1
Pesticide (kg ha^{-1})	0.4[c]	3.92 [b]	1.57
Sub Total			***288.77***
Field operations	**Consumption (l ha^{-1})**	**GWP 100 (kg CO_{2eq} l^{-1})**	**GHG emissions (kg CO_{2eq})**
Plough with disc harrows	7[e]	2.6[d]	18.2
Sowing+ Fertilization	8 [e]	2.6[d]	20.8
Weeding with hydraulic sprayers	4 [e]	2.6[d]	10.4
Harvesting + Baling	12 [e]	2.6[d]	31.2
Transport of workers (5km)	1.5 [e]	2.6[d]	3.9
Transport of product (5 km, go empty)	2 [e]	2.6[d]	5.2
Transport of product (5 km, return full)	2.5 [e]	2.6[d]	6.5
Sub Total			***96.2***
Emission from fertilisers	N_2O emissions (kgN_2O)	GWP 100 (kg CO_{2eq} N_2O^{-1})	GHG emissions (kg CO_{2eq})
Emissions from N fertiliser (45+78.2 Kg N)	3.1[h]	298[g]	923.8
Sub Total			***923.8***
Total			***1321.1***

[a] Rosado (2009)
[b] Kramer et al. (1999)
[c] Abrahao et al. (2017)
[d] Mäkelä (2002)
[e] IDAE (2005)
[g] IPCC (2006)
[h] Based on the conversion factor for N fertiliser to N_2O emissions of 2.55% (Rajaniemi et al. 2011)

The trees were assumed to require fertilization at the rate of 125g of NPK (7:21:21) per plant (Gonzalez-Garcia et al. 2013). During the first year, 20% of the trees were assumed to require replacement due to mortality without additional fertilizer being applied. Pruning was assumed to be required every eight years and cork debarking every nine years, but this was done manually and therefore not considered to be a GHG emission source. Usually in a cork oak plantation, vegetation clearing is undertaken every four to five years but as wheat is cultivated in the area between the trees, this operation was not included here. A petrol chainsaw was considered for pruning and thinning operations.

Table 17. Source of greenhouse gas emissions for cork oak agroforestry management

Input production	**Quantity**	**GWP 100 (kg CO_{2eq} kg^{-1})**	**GHG (kg CO_{2eq})**
Plants	200	-	-
Fertiliser kg N	1.75	2.66 b	4.6
Sub Total			***4.6***
Field operations	**Consumption ($l\ ha^{-1}$)**	**GWP 100 (kg CO_{2eq} l^{-1})**	**GHG emissions (kg CO_{2eq})**
Clearing	6.8[c]	2.6[d]	17.7
Ripping	20[c]	2.6[d]	52
Plough with disc harrows	9.8[c]	2.6[d]	25.48
Planting + Fertilization	Manual	-	-
Replanting	Manual	-	-
Transport of workers (5km)	1.5[c]	2.6[d]	3.9
Sub Total			***99.08***
Tree operations	Consumption ($l\ tree^{-1}$)	GWP 100 (kg CO_{2eq} l^{-1})	GHG emissions (kg CO_{2eq})
Pruning (depends on tree density)	0.26[e]	2.3[d]	0.6*
Thinning (depends on tree density)	0.10[e]	2.3[d]	0.2*
Debarking	Manual	-	-
Sub Total			***0.8***
Emission from fertilisers	N_2O emissions (kg N_2O)	GWP 100 (kg CO_{2eq} N_2O^{-1})	GHG emissions (kg CO_{2eq})
Emissions from N fertiliser (1.75 kg N)	0.04[h]	298[g]	13.29
Sub Total			***13.29***
Total			**117,77**

[a] Gonzalez-García et al. (2013)
[b] Kramer et al. (1999)
[c] IDAE (2005)
[d] Mäkelä (2002)
[e] CAOF (2010)
[g] IPCC (2006)
[h] Based on the conversion factor for N fertiliser to N_2O emissions of 2.55% (Rajaniemi et al. 2011).
*The values for pruning and thinning operation are presented per tree as the total value depend on the tree density.
**The values are presented per hectare.

Kramer et al. (1999) reported 0.241, 2.66 and 3.96 kg $CO2_{eq}$ kg^{-1} for the production of wheat seeds, nitrogen fertiliser, and pesticides respectively. GHG emissions derived from the use of nitrogen fertiliser were added to the field operation emissions, since approximately 2.55% of N-fertiliser is converted to N_2O (Rajaniemi et al. 2011).

Modelling with Yield-SAFE

The Yield-SAFE model is a process-based dynamic model for predicting resource capture, growth, and production in forestry, agroforestry and agricultural systems (van der Werf et al. 2007, Palma et al. 2016, 2017c). The model can be used to estimate carbon sequestered by the tree and crop biomass and the soil using an application of the RothC model within Yield-SAFE (Palma et al. 2017b). The carbon sequestered was estimated assuming that 50% of the biomass was carbon. Biomass in this study included above-ground biomass (AGB) and below-ground biomass (BGB) that is estimated using a root-to shoot ratio of 0.43 and 0.31 for cork oak and wheat respectively (Siddique et al. 1990; Palma et al. 2014) but excluded both systems output products including wood from pruning, cork debarked, wheat grain and straw. Tree leaves and roots and wheat roots after harvesting were included as inputs for the soil carbon model. The period of simulation was 80 years and the weather data used for the simulations was extracted from Clipick (Palma, 2017a) which uses data from the KNMI regional atmospheric climate model RACMO (version 2.2) previously tested in the country (Palma et al. 2018). Cork oak and wheat parameter sets for Yield-SAFE were taken from Palma et al. (2014, 2017c).

Carbon Balance estimation

The estimation of the Carbon Balance of the wheat and agroforestry systems was calculated as the difference between: 1) the amount of GHG emissions with GWP (CO_{2eq}) emitted by different activities and products used during the growth process and 2) the amount of carbon sequestered in the above- and below-ground biomass and in the soil (**Figure 16**). The CO_{2eq} emissions released from soil biota were included as GHG emission (soil respiration). Results were expressed in Mg of CO_{2eq} Mg of wheat grain^{-1}.

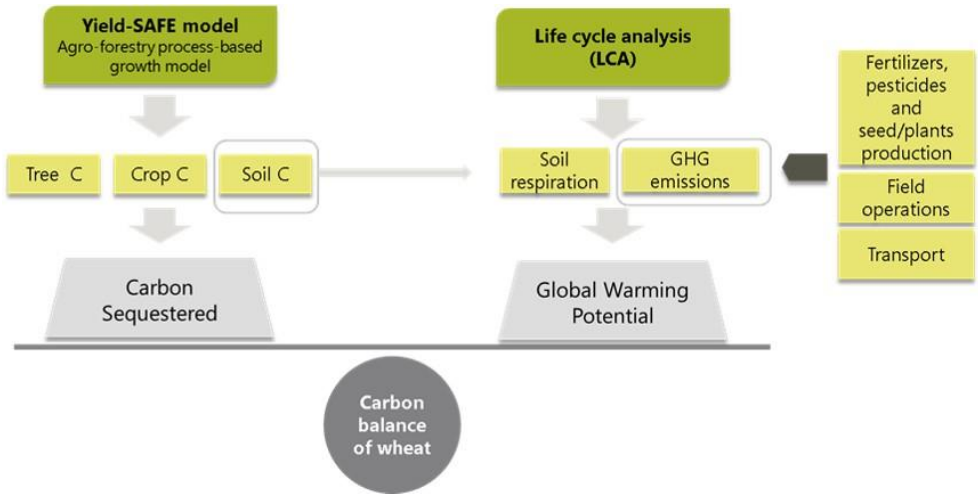

Figure 16. Methodology used to estimate the Carbon Balance of a functional unit (Mg of CO_{2eq} Mg of wheat grain^{-1}) by comparing the estimation of the Yield-SAFE model of the carbon sequestered by the tree, crop and soil components and the greenhouse gas emissions derived from fertilizers, pesticides and seed/plants production, field operations, transport and soil respiration.

Results

Predicted yields of the Yield-SAFE model

The average wheat yield over 80 years for the wheat monoculture was predicted by Yield-SAFE to be 1.73 Mg ha^{-1}year^{-1} (excluding the fallow years) (**Figure 17**). This yield was slightly higher than the expected wheat yield in the country that is of 1.4 Mg ha^{-1} year^{-1} (Almeida and Maçâs 2016). The agroforestry wheat on a per hectare crop basis, compared to the monoculture wheat, had similar yields during the first part of the simulation period (from year 1 to year 30), slightly lower yields during the middle part of the simulation period (from year 30 to year 50), and much lower yields during the last part of the simulation period (form year 50 to year 80) (**Figure 17**). However, on average, wheat production on the agroforestry system was of 1.53 Mg ha^{-1}year^{-1}. The accumulated biomass in the tree component was approximately 0.650 Mg tree^{-1} and this was similar to the above-ground biomass for trees of the same age in similar conditions in Portugal (Palma et al. 2014).

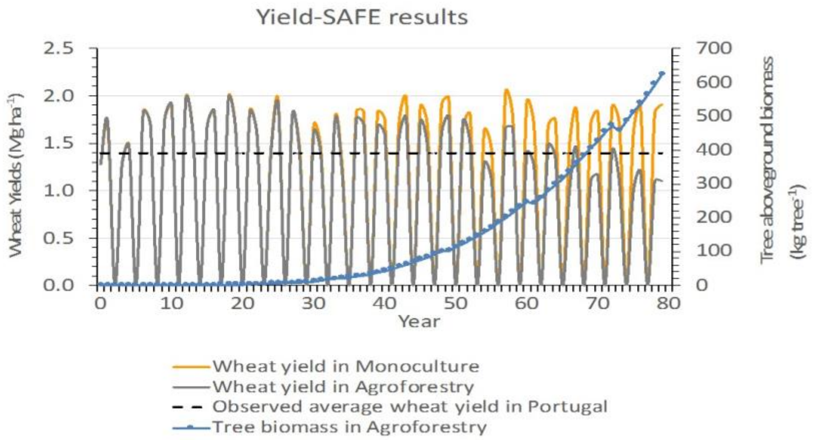

Figure 17. Tree growth and wheat yields simulated by the Yield-SAFE model for a wheat monoculture (Wheat yield in monoculture) and an agroforestry system (Wheat yield in Agroforestry and Tree biomass in Agroforestry) combining wheat and cork oak trees in Portugal compared to the observed average yield in Portugal for the period 1990-2011 by Almeida & Maças in 2016 (Observed average wheat yield in Portugal).

Carbon Balance

The predicted GWP for wheat of 0.81 Mg CO_{2eq} Mg of wheat^{-1} was close to the reported in Rosado (2009) of 1.08 Mg CO_{2eq} Mg of wheat^{-1} for lower average yields. For cork oak (Gonzalez-García et al. 2013) reported GWP potential from forest operation of 1.64 Mg CO_{2eq} ha^{-1} for a cork oak stand of 100 tree ha^{-1} whilst in this study, the GWP from forest operations (including fertiliser use) were 0.49 Mg CO_{2eq} ha^{-1} although this was for a final tree density of 35 tree ha^{-1} and did not include the vegetation clearing operation that occurred every 3-4 years in Gonzalez-García et al. (2013). In terms of carbon sequestration of the tree component of the system, in this study above-ground biomass was estimated in 1.19 Mg CO_{2eq} in year 80 (0.325 Mg C tree^{-1}). This is similar to Palma et al. (2014) who used Yield-SAFE and the same carbon sequestration method for a slightly higher final tree density (50 trees ha^{-1}) agroforestry system in Portugal and reported a cork oak tree carbon content of 1.32 Mg CO_{2eq} (0.362 Mg C tree^{-1}) in year 80.

GHG emissions from wheat field operations, and fertiliser, seed and pesticide production were the main contributions to the GWP identified (GHG emissions in **Figure 18**). For agroforestry were quantified in the range of 0 and 1.34 Mg CO_{2eq} ha^{-1} depending on crop and tree field operations (GHG emissions in **Figure 18**) and of between 0 and 1.4 Mg CO_{2eq} ha^{-}1 for the monoculture system depending if wheat was cultivated or not. This source of emission was slightly lower for agroforestry due to the fact crop area was also lower (10% less). Respiration from the soil biota activity was also a source of GHG emissions (Soil respiration in **Figure 18**) and was related to the crop yield: the soil can act as a sink or as an emitter of carbon depending on the quantity of plant material accumulated within it during the year. In the wheat monoculture system, even in wheat cultivation years, the input of plant material was not sufficient to offset soil carbon loss by respiration and soil carbon content decreased by 20% over the simulation period. By contrast, in the agroforestry system, additional input from trees (roots and leaves) allowed the soil to increase the quantity of carbon stored and therefore act as a carbon sink, and this especially from years 35 onwards (**Figure 18**). The effects of the gains or losses in terms of carbon in soil are reflected in the total soil carbon content (**Figure 18**). The tree component (Tree carbon in **Figure 18**) of the agroforestry system was the largest carbon sink for most of the simulation period. During the thinning years the amount of carbon stored in the tree component was assumed to be neutral i.e. there are neither losses nor gains of carbon in the tree component. This is because in thinning years the tree density is reduced, and the remaining tree's do not compensate the carbon losses derived. As these outgoing trees continue to store the carbon even outside the system it is considered that during these years there is neither positive nor negative tree growth.

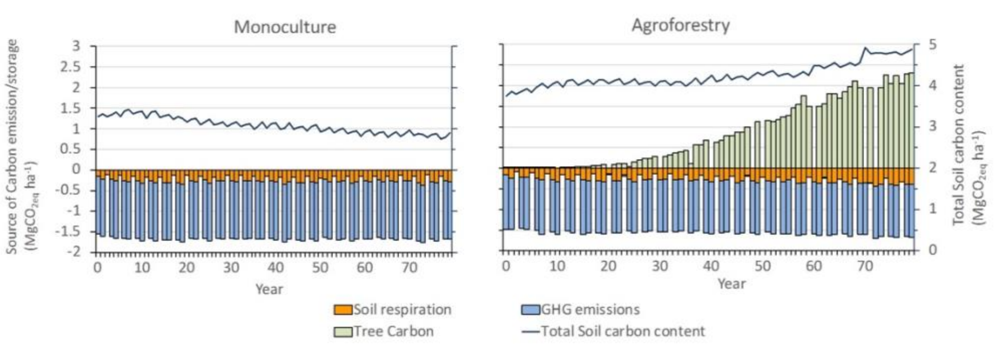

Figure 18. Carbon sources of emissions and storage (in $MgCO_{2eq}$ ha^{-1}) for a Monoculture and Agroforestry management systems for wheat production including emissions from soil biota respiration (Soil Respiration), from field operations and fertiliser, seed and pesticide production (GHG emissions) and carbon sequestered by tree (Tree Carbon) and stored in soil (Total Soil carbon content).

Agroforestry for Food Sustainable intensification

The second step of this study was to assess the balance between the GHG emitted and the carbon sequestered during production of wheat in the agroforestry and monoculture systems. If the same quantity of wheat is produced on the same area of land, with reduced carbon emissions, this can be considered to satisfy the SI process. The average yield in the agroforestry system (1.53 Mg of wheat ha^{-1} $year^{-1}$) was found to be slightly lower over the 80-year simulation period than the monoculture yield (1.73 Mg of wheat ha^{-1} $year^{-1}$). Results showed that in the monoculture management option, the production of 1 Mg of wheat was associated with a negative carbon balance result of 1 Mg CO_{2eq} (**Figure 19**)

In agroforestry, the production of 1 Mg of wheat was linked to an initial negative carbon balance of around 1 Mg CO_{2eq} whilst the trees were small. But as the trees grew over time, the carbon balance improved and by year 50 became positive, suggesting that agroforestry, when the trees become mature, could have a positive carbon balance of 1 Mg CO_{2eq} for the production of each Mg of wheat (**Figure 19**).

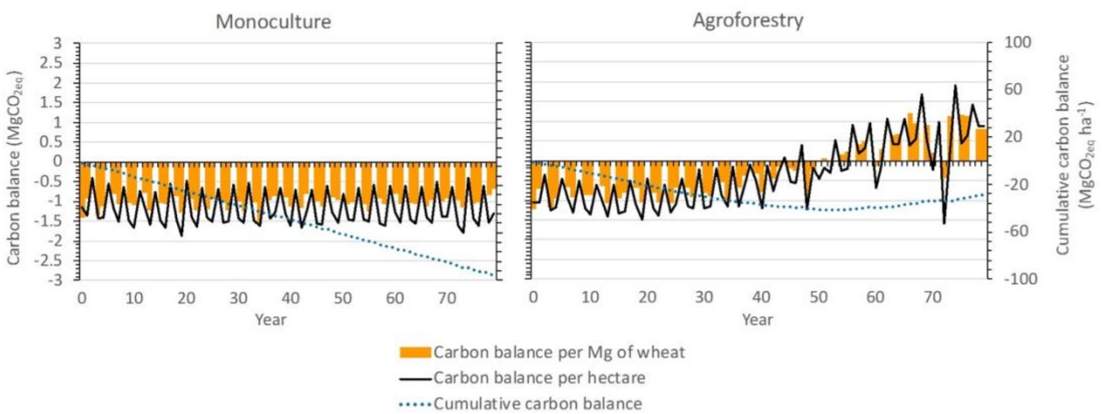

Figure 19. Differences between carbon emissions and carbon sequestered per ton of wheat produced (Carbon balance per Mg of wheat), per hectare of system occupied by Monoculture or Agroforestry (Carbon balance per hectare) and the cumulative values (Cumulative carbon balance) for two management option for wheat production in Portugal (Monoculture and Agroforestry).

Discussion

The methodology presented here integrates process-based modelling and LCA so that changes in practices that could lead to sustainable intensification can be evaluated. This is done by comparing the GHG emissions and carbon sequestration trade-off of a functional unit, in this study, the production of 1 Mg of wheat. Furthermore, the carbon balance integrates dimensions that can be used to consider the three main challenges for SI referred to by the FAO, which include: 1) food security, 2) climate change mitigation and 3) land occupation. Producing results in terms of a carbon balance per Mg^{-1} crop ha^{-1} $year^{-1}$ helps to provide an approach that can be used to make the SI concept more workable, an aspect of the concept that has been challenging (Petersen and Snapp 2015).

In this study, the results of the carbon balance for wheat under two different land management options has confirmed that the agroforestry option provided an improvement over the monoculture because it: 1) produced similar yields (ensures food security); 2) helped with climate change mitigation (positive carbon balance after year 50) and 3) avoided the need to increase the area of land occupied by agriculture as per hectare yields were similar to the monoculture option.

In agroforestry systems the integration of woody vegetation together with wheat allows the system to make more efficient use of natural resources such as light and water resulting in higher land equivalent ratios that can be achieved by growing trees and crops separately (Graves et al. 2010). This may be due to complementarity in resource use or sometimes because trees may help to retain natural resources such as soil and water, increasing the amount of energy accumulated and the provisioning of ecosystem services provided (Crous-Duran et al. 2018). However, these findings are also contested (Cubera et al. 2009, Rivest et al. 2011, Torralba et al. 2016). Nevertheless, the algorithms recently implemented in Yield-SAFE (Palma et al 2016) attempt to account for the effect that trees have on buffering minimum and maximum temperatures whilst reducing wind speeds. This leads to reduced water losses due to reduced evapotranspiration, enabling an extension of the growing season, or at least as in this study, maintain yields although tree competition.

The agroforestry system here was found to reduce GHG emissions directly. In fact, even if the combination of the two activities (forestry and agriculture) increased the number of field operations required, the reduction of area dedicated to crop and the fertiliser used reduced the total GHG emitted for the crop area from 1.4 Mg CO_{2eq} to 1.27 Mg CO_{2eq} ha^{-1}. However, here, after year 50 of the rotation, a positive carbon balance was nevertheless achieved and this was due to the incorporation of the trees that after a certain amount of time were able to numerically offset the GHG emissions associated with both activities whilst maintaining a similar level of crop production.

The increase in the demand for food has been met either by increasing fertiliser, machinery use, or genetic improvement (intensification), or by increasing the area of land occupied by agriculture (extensification). Both strategies have large impacts on GHG emissions and if badly managed can lead to an impoverishment of soils and in a reduction of yields as a consequence, as was the case in the silvo-arable *montado* in Portugal (Pinto-Correia 1993). Agroforestry is a strategy that can help to increase food production without requiring the conversion of new land. As noted by the FAO, this is an important aspect of intensification.

Furthermore, agroforestry helps provide important environmental benefits like reducing soil erosion (Nair et al. 2007a), nitrate leaching (Palma et al 2007b, Jose 2009), net greenhouse gas emissions (Godfray et al. 2012) and improve biodiversity conservation (Torralba et al. 2016), soil enrichment (Graves et al. 2015) and enhance climate change mitigation by sequestering more carbon in soils (Cardinael et al. 2017).

Whilst, not evaluated in this study, these benefits strengthen the case for the promotion and implementation of agroforestry systems in Europe as SI practices.

Compared to crop monocultures of trees and crops, agroforestry systems can help to ensure farm profitability as crop yields and tree growth are similar or even higher (García de Jalón et al. 2018), enhance financial security as production is diversified, and diversify and stimulate rural economics through new product streams.

In the case of Portugal, this work suggests that implementation of agroforestry systems instead of crop monocultures for wheat production is preferable, because this land use option, when mature (from year 50 onwards), could provide a net carbon sequestration rate of around one Mg of CO_{2eq} for every Mg of wheat produced. And even more when is stated that considering the existent soil and climate conditions other agricultural uses would of difficult implementation (Pinto-Correia 1993). In Portugal in 2011, the area of wheat production was 276,000 ha and the average wheat yield was about 1.4 Mg ha^{-1} producing a total of 386,400 Mg of wheat per year (Almeida & Maçâs 2016). A change of production from crop monoculture to agroforestry could result in substantial carbon sequestration in the near future to offset the GHG emissions associated with wheat production.

The carbon balance method as presented here appears to be a useful indicator for evaluating SI practices. In this study, the method was applied to wheat production in Portugal as this is a significant crop in the country and globally. However, the carbon balance method could be used for evaluating other provisioning ecosystem services, including other crops, meat, timber, cork, nuts or fruit. The method could also be applied to forestry and orchard systems as the base methodologies used (LCA approach and Yield-SAFE model) are compatible.

Conclusion

This study compares the carbon emissions and carbon sequestration to produce a carbon balance of the product being assessed. Applied to food production the method here enables the SI of different management options to be compared. The results in Mg CO_{2eq} Mg $food^{-1}$ applied for the same area (1 ha of system) and time (1 year) facilitate this comparison. Positive values represent net carbon sequestration whilst negative values represent net carbon emissions. The SI potential was analyzed for wheat production for two different management options, a crop monoculture and an agroforestry system. The crop monoculture had a negative carbon balance during the entire simulation (80 years) of around 1 Mg CO_{2eq} Mg $wheat^{-1}$. Under a cork oak agroforestry system, the carbon balance was positive from year 50 onwards and for every Mg of wheat produced 1 Mg of carbon dioxide equivalent was sequestered confirming that agroforestry could be used as a SI practice. The carbon balance method presented here is a useful indicator for evaluating SI practices and whilst the method was here applied for wheat production in Portugal, it could be easily used for evaluating other provisioning ecosystem services or combinations of ecosystem services in agroforestry and non-agroforestry systems.

Acknowledgments

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Chapter 6 | Whole system valuation of arable, agroforestry and tree-only systems at three case study sites in Europe

Based on Giannitsopoulos M, Graves AR, Burgess PJ, **Crous-Duran J**, Moreno G, Herzog F, Palma JHN, Kay S, García de Jalón S. Whole system valuation of arable, agroforestry and tree-only systems at three case study sites in Europe. Journal of Cleaner Production 269C, 122283. DOI: 10.1016/j.jclepro.2020.122283

Abstract

There is an increasing demand to study the long-term effects of land use from both local farm and wider societal and environmental perspectives. This study applied an approach to evaluate both the financial profitability of arable, agroforestry, and tree-only systems and the wider societal benefits over a period of 30-60 years. The bio-physical inputs and yields from the three systems were modelled for three case study sites in the United Kingdom, Spain, and Switzerland, using a tree and crop simulation model called Yield-SAFE. A bio-economic model called Farm-SAFE was then used to compare the financial (EAV_F) and economic (or societal) equivalent annual values (EAV_E) by including monetary values for five environmental externalities: carbon dioxide emissions, carbon sequestration, soil erosion by water, and nitrogen and phosphorus balances. Across the three case studies, arable farming generated higher farm incomes than the agroforestry or tree-only systems, but the arable systems also created the greatest environmental costs. By comparison the agroforestry and tree-only systems generated lower CO_2 emissions and sequestered more carbon. Applying monetary values to the environmental externalities meant that the EAV_E of the agroforestry and tree-only systems were greater or similar to that for the arable system in the UK case study. In Spain, the slow predicted growth of the trees meant that, even after including the environmental externalities, the arable system created greater societal benefit than the agroforestry and tree-only systems. In Switzerland, including the environmental externalities increased the attraction of the tree-only system, but the high subsidies for arable and agroforestry systems meant that the EAV_E for the agroforestry and arable systems were the most attractive from a farmer's perspective. A breakeven analysis was used to determine the environmental externality values at which the agroforestry and tree-only systems produced the same societal return as the arable system in each case study. In the UK, a carbon price of €16 (t CO_2)$^{-1}$ allowed the EAV_E of the agroforestry system to attain parity with the arable EAV_E. In both the UK and Spain, an environmental nitrogen cost of €3-6 (kg N)$^{-1}$ was sufficient for the EAV_E of the agroforestry and tree-only systems to match those of arable farming. Because trees on farms provide "economies of multifunction" for environmental benefits, the breakeven values will be less if environmental benefits are considered together as packages. The described

approach provides a method for governments and others to examine the cost effectiveness of new agri-environment measures.

Keywords

Ecosystem services, land-use, carbon sequestration, nitrogen leaching, soil erosion

Introduction

Increased agricultural production per unit of land area and per unit labour was achieved in 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 (European Parliament 2019; UK Government 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 (Blaser et al. 2018; Hernández-Morcillo et al. 2018; Kay et al. 2019b). 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 include an increase in biodiversity relative to monoculture crop or forest systems (Torralba et al. 2016; Blaser et al. 2018), and improved soil conservation relative to monoculture arable systems (Herzog 2000).

The uptake of agroforestry practices in Europe is often constrained by the low profitability of agroforestry relative to tree-only and arable-only systems (García de Jalón et al. 2017a). 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 encourage increased uptake of 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. 2017b) or a single area (Ovando et al. 2017). Kay et al. (2019a) 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 study is to develop 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.

Materials & Methods

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 18**) 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* silvo-pastoral 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 18**). The next four stages of the method were to: (i) simulate the bio-physical 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 18. Arable, agroforestry, and tree-only systems were compared at each of three case study sites.

	Bedfordshire UK	**Extremadura ES**	**Schwarzbubenland CH**
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 trees ha^{-1} [c] Year 45-60: 250 trees ha^{-1}	*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)

Bio-physical simulation

Simulated daily temperature, solar radiation and rainfall data for each site were obtained using the CliPick tool (Palma 2017a). 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 bio-physical 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. 2017b) 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.

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 by using management handbooks and working with farmers and advisors, and the resulting information was stored together to ensure a consistent financial dataset for each site. The crop input costs, along with tree data such as establishment, weeding and pruning (Appendix 5; Table B1 and Table B3) and the levels of governmental support (Appendix 5; Table B2) 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 9**) 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:

$$\mathrm{NPV_F} = \sum_{t=0}^{n} \left(\frac{(R_t - VC_t - AFC_t)}{(1+i)^t} \right) \qquad \text{Equation 9}$$

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 10**:

$$\mathrm{EAV_F} = \mathrm{NPV_F} * \left(\frac{(1+i)^n}{(1+i)^n - 1} \right) * i \qquad \text{Equation 10}$$

Modelling the environmental externalities

The environmental externalities were modelled using the approach described by (García de Jalón et al. 2017b) 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.

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 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 11**). This analysis did not consider CH_4 and N_2O.

$$Emi.CO_2 = M_F + M_P + M_M + F_C + F_F + F_S + F_H \qquad \text{Equation 11}$$

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.

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** 12). 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.

$$Total\ Seq. C = (Timber + Branchwood + Roots) + Soil \qquad \text{Equation 12}$$

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 13**):

$$A = RKLSCP \qquad \text{Equation 13}$$

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 19**). 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 (Prasuhn et al. 2007; Panagos et al. 2014; Panagos et al. 2015b; Panagos et al. 2015d; Panagos et al. 2015a).

Table 19. 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, SP	518.5	0.03	1.50	T:0.40	G:0.20*		1
Schwarzbubenland, CH	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 (Wischmmeireeier and Smith, 1978)

The dynamic change in the cover-management factor (C_t) in year *t* was calculated for each system using **Equation 14**:

$$C_t = Cov_{crop,t}C_{crop} + Cov_{tree,t}C_{tree} \qquad \text{Equation 14}$$

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. 2015d). Our approach considered the distance between tree lines as in (Palma et al. 2007a) and the changes in land cover fraction over time as a result of tree canopy growth.

Nitrogen balance

As described in (García de Jalón et al. 2017b) the annual nitrogen balance (N_{bal}; units: kg N ha^{-1} y^{-1}) of each land use system was determined using Palma et al. (2007c) and (Feldwisch et al. 1998) (**Equation 15**):

$$N_{bal} = N_{fert} + N_{Adep} + N_{fix} + N_{min} - D - V - U - I \qquad \text{Equation 15}$$

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 5.

Phosphorus balance

Annual phosphorus balance ($P_{bal;}$ units: kg P ha^{-1} y^{-1}) was calculated from **Equation 16** which shows the P inputs and outputs considered in the analysis.

$$P_{bal} = P_{fert} + PA_{dep} - U \qquad \text{Equation 16}$$

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 5. 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).

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 $\mathrm{NPV_F}$ (see Equation 9) 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 ($\mathrm{NPV_E}$; **Equation 17**) was determined as:

$$\mathrm{NPV_E} = \sum_{t=0}^{n}\left(\left(\frac{(R_t - VC_t - FC_t)}{(1+i)^t}\right) + \left(\frac{EE_t}{(1+j)^t}\right)\right) \qquad \text{Equation 17}$$

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 $\mathrm{NPV_E}$, the economic EAV ($\mathrm{EAV_E}$) was calculated as in **Equation 10**.

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 $\mathrm{EAV_E}$ of the agroforestry and tree-only systems matched the $\mathrm{EAV_E}$ 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 $\mathrm{EAV_E}$ relative to systems that sequester carbon. The value of $\mathrm{EAV_E}$ 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.

Results

Bio-physical 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 20**). 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 20. 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

Financial analysis

In the UK system, excluding 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 21**), 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 21**).

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 21**) 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 21**).

In the Swiss system without grants, the tree-only and agroforestry systems resulted in a negative NPV_F and EAV_F, highlighting their reliance on grants (due to high labour and machinery costs). 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 5; Table B2) the Swiss arable and agroforestry systems were the most profitable (**Table 21**). 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 21**). 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.

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 3). 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 3). 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 3).

Table 21. 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	Years	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

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 20**). 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 20**). 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.

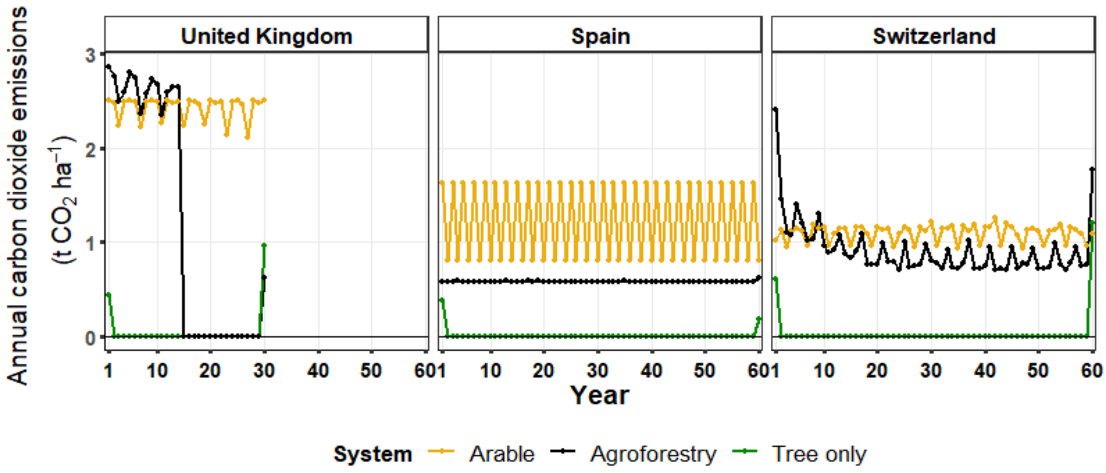

Figure 20. Modelled annual CO_2 emissions for the arable, agroforestry, and tree-only for UK, Spain, and Switzerland over 30, 60, and 60 years respectively.

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 silvo-pastoral 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 21**). 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 21**). The carbon storage in the UK tree-only system was 104 t C ha^{-1} in 30 years.

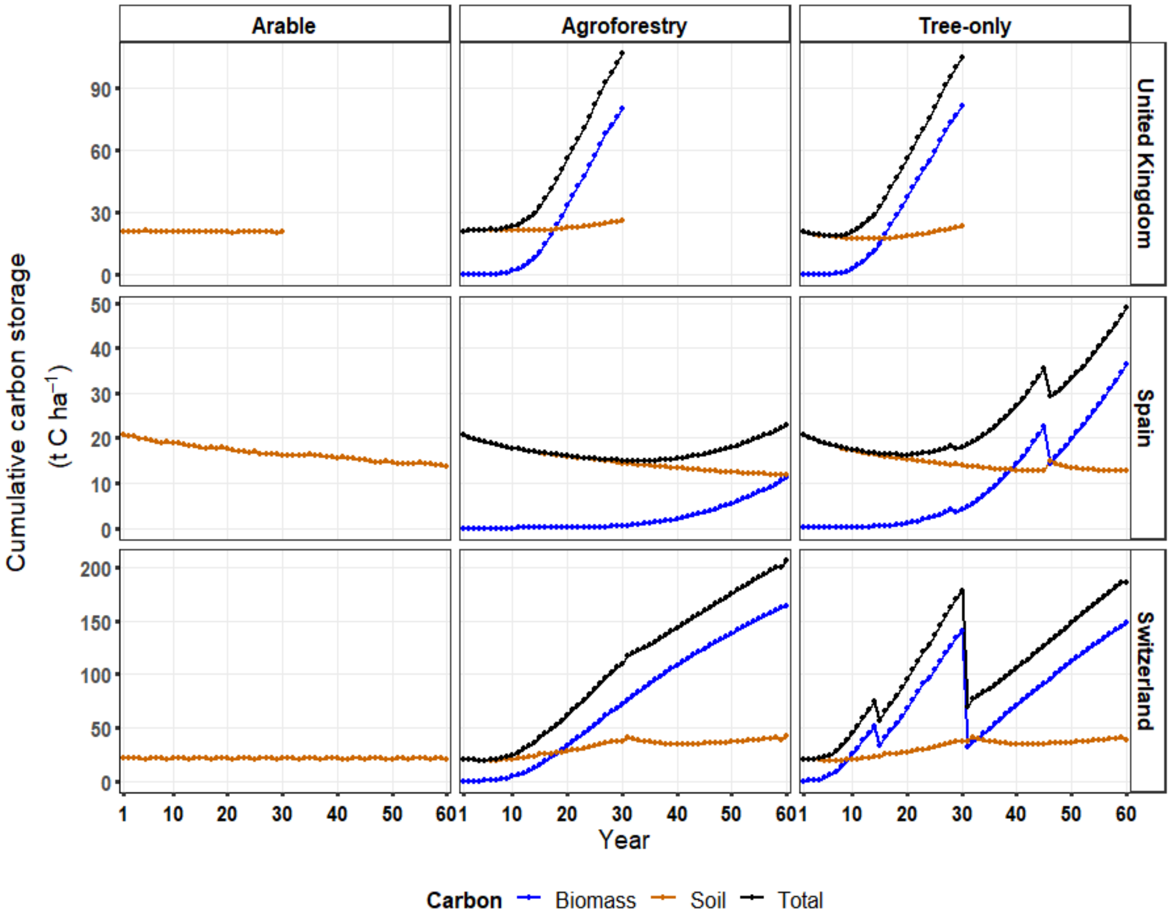

Figure 21. Modelled carbon storage (t C ha^{-1}) as: biomass (above and belowground), soil (which also includes fallen leaf carbon) and in terms of total carbon (biomass plus soil). In the arable system, the soil carbon is the same as the total carbon as the biomass carbon was assumed to be zero. Note 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 21**). 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 21**).

The cumulative net carbon sequestration of each system (**Figure 22**) was calculated by combining the cumulative net CO_2 emissions (Appendix 5; Figure C1) with the cumulative sequestered carbon (**Figure 21**). 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.

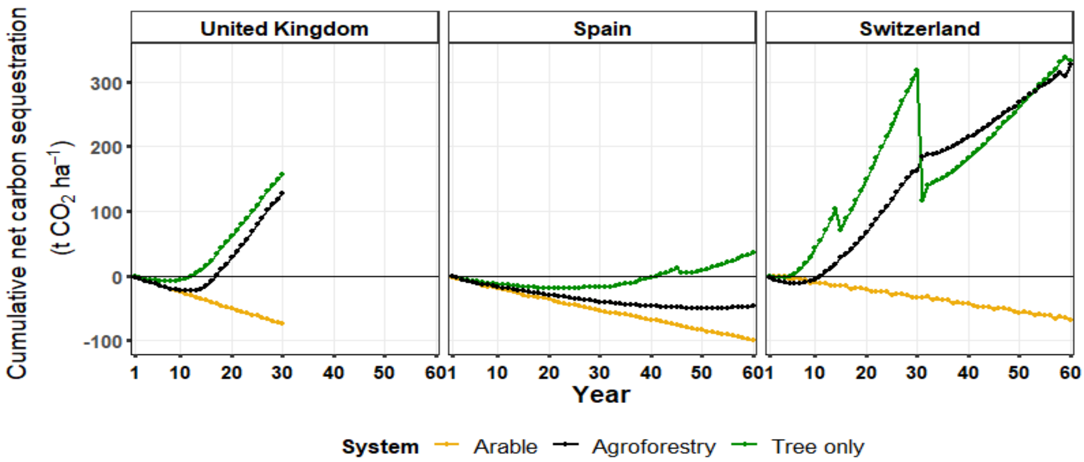

Figure 22. Modelled cumulative net carbon sequestration for the arable, agroforestry and tree-only systems for UK, Spain, and Switzerland over 30, 60, and 60 years respectively.

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 23**; Appendix 5: Figure C2). 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 5: Figure C2).

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 5: Figure C2). 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 22**). 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 22**).

Table 22. 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

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 23**). 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 23**). 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 23**). 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}).

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.

Table 23. 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

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 24**). By contrast, in Switzerland without grants, the tree-only system 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 24**).

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 24.**; Appendix 5; Table B4). 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 24**; Appendix 5 Table B4).

Table 24. 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 23 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 3. 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 3).

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 23**). 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 23**; 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 23**). 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 19**; Appendix 3).

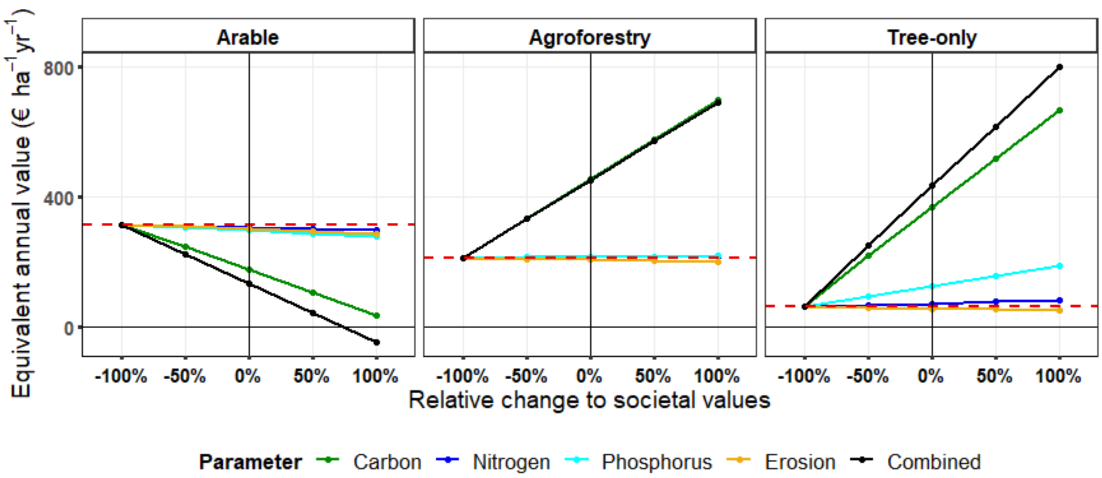

Figure 23. Sensitivity analysis on the equivalent annual value for the United Kingdom per land use system (when not accounting for grants) as affected by the value of environmental externalities relative to the default prices of €57.1 (t CO_2)$^{-1}$, €0.20 (kg N)$^{-1}$, €1.58 (kg P)$^{-1}$ and €6.4 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 3). 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 3).

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 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 24**; Appendix 5: Table B4). 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 24**; **Figure 23**; Appendix 5: Table B4).

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 24**). 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 24**). 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 silvo-arable 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 24**).

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 24**). 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 in the UK mean that the arable area would receive €235 ha^{-1} y^{-1}, whereas tree planting at a relatively low tree density of 156 ha^{-1} would not be eligible for government subsidies. Although the UK government (HM Government 2018) argues that public money provided to farmers should be targeted for the provision for non-market public services, the current grant system operates in the opposite direction favouring arable production and creating a disadvantage for low density tree planting. If agricultural subsidies in the UK were directly related to the public benefits, then the tree-only and the agroforestry systems could be more financially profitable to the farmer than 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 benefits and costs of the three systems are relatively similar (ranging from -€154 ha^{-1} y^{-1} for arable to €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 three systems would be broadly similar (€-45 ha^{-1} y^{-1} to €51 ha^{-1} y^{-1}).

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 related to recreation and biodiversity would further strengthen the attraction of tree planting and management.

Another limitation of the analysis is the assumption that farmers will automatically change from one land use system to another directly in response to the most profitable EAV. In practice, land use decisions are determined using a wide range of criteria (García de Jalón 2018a). 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. By contrast, some farmers will engage in land use change for non-financial reasons.

The Yield-SAFE bio-physical model and the Farm-SAFE economic models have been used in previous studies to predict the bio-physical 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 2017b). 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.

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.

Acknowledgements

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Chapter 7 |Other scientific publications

Introduction

During his doctoral studies, the candidate had the chance to collaborate on other publications that emerged from the AGFORWARD project. These collaborations were mainly based on analysing aspects of agroforestry systems from a social, ecological or economic point of view and at a different scale proposed for thesis which takes a plot scale perspective.

For Moreno et al (2017) the candidate helped by providing information on the Portuguese *montado* for the review based on literature and stakeholder knowledge of the structure, components, management practices, and the ecosystems services provided by ten agroforestry systems considered of high nature and cultural value (HNCV). In Fagerholm et al (2019), the candidate led the Portuguese case study by interviewing 270 residents inhabiting a *montado* landscape near Montemor-o-Novo as part of a European scale study to understand how the benefits of ecosystem services are perceived and mapped by residents in different landscapes. In Rolo et al (2020) the results of participatory research with ten stakeholder groups in Europe trying to find solutions for the economic and ecological viability of High Nature and Cultural Value Agroforestry systems (HNCV) are analysed. The candidate co-organized the Portuguese stakeholder meeting, collected the answers and developed the report (Crous-Duran et al. 2014).

The candidate helped to upgrade the bio-physical model Yield-SAFE first by implementing the soil carbon model RothC, a model simulating soil organic carbon turnover (Palma et al. 2017b) and then by updating the model with new algorithms to calculate the livestock carrying capacity, the fruit (acorn) production, the effect of heat stress on livestock weight gain and the canopy effect on air temperature and wind speed, applied the updated version on the *montado* system (Palma et al 2016).

In terms of bio-economic modelling, the candidate helped to develop a new model (Forage-SAFE) able to examine the environmental benefits of introducing additional trees within wood pasture systems on farm profitability of a *dehesa* system in South-western Spain (García de Jalón et al. 2018a) and also helped to determine the capacity of trees to remove air pollutants by dry deposition and estimate its annual economic value in the Basque Country, Spain (García de Jalón et al. 2019).

Rural landscapes can also be enhanced with agroforestry systems as these can provide more and better ecosystem services compared to conventional agriculture. In Kay et al. (2018a), the candidate helped to compare the provision of ecosystem services of agroforestry and non-agroforestry landscapes in several regions in Europe and specially for the Mediterranean area and the *montado* and *dehesa* systems. This suggested that that regulating ecosystems services were improved in all agroforestry landscapes while the results for provisioning services were inconsistent. In Kay et al. (2018b) the candidate helped to develop a novel and spatially explicit model to assess and quantify bundles of Provisioning and Regulating ES provided by landscapes with and without agroforestry systems. In Kay et al. (2019a) the candidate took part in the study for quantifying the economic performance of these ES in 11 contrasting European landscapes dominated by agroforestry land use compared to business as usual agricultural practice. Finally, the candidate helped by combining scientific and technical knowledge to evaluate nine environmental pressures in terms of ecosystem services in European farmland and assessed the carbon storage potential of suitable agroforestry systems proposed by regional experts (Kay et al. 2019b).

During the development of this thesis the candidate also helped in other scientific publications. The aim of the study in den Herder et al. (2017) was to quantify and map the distribution of agroforestry in the European Union using Land Use and Land Cover data (LUCAS) data while in Gidey et al. (2019) the candidate offered support to use the Yield-SAFE model for assessing coffee production in Ethiopia under coffee monoculture and coffee agroforestry systems under current climate and two different future climate change scenarios.

Social-cultural perception

Moreno G, Aviron S, Berg S, **Crous-Duran J**, Franca A, de Jalón SG, Hartel T, Mirck J, Pantera A, Palma JHN, Paulo JA, Re GA, Sanna F, Thenail C, Varga A, Viaud V, Burgess PJ (2017). Agroforestry systems of high nature and cultural value in Europe: provision of commercial goods and other ecosystem services. Agroforestry Systems. DOI: 10.1007/s10457-017-0126-1

Abstract: Land use systems that integrate woody vegetation with livestock and/or crops and are recognised for their biodiversity and cultural importance can be termed high nature and cultural value (HNCV) agroforestry. In this review, based on the literature and stakeholder knowledge, we describe the structure, components and management practices of ten contrasting HNCV agroforestry systems distributed across five European bioclimatic regions. We also compile and categorize the ecosystem services provided by these agroforestry systems, following the Common International Classification of Ecosystem Services. HNCV agroforestry in Europe generally enhances biodiversity and regulating ecosystem services relative to conventional agriculture and forestry. These systems can reduce fire risk, compared to conventional forestry, and can increase carbon sequestration, moderate the microclimate, and reduce soil erosion and nutrient leaching compared to conventional agriculture. However, some of the evidence is location specific and a better geographical coverage is needed to generalize patterns at broader scales. Although some traditional practices and products have been abandoned, many of the studied systems continue to provide multiple woody and non-woody plant products and high-quality food from livestock and game. Some of the cultural value of these systems can also be captured through tourism and local events. However there remains a continual challenge for farmers, landowners and society to fully translate the positive social and environmental impacts of HNCV agroforestry into market prices for the products and services.

Keywords: Wood pastures, Bocage, *Dehesa*/*montado*, Parklands, Biodiversity, Provisioning Ecosystem Services, Regulating services, Cultural service

Fagerholm N, Torralba M, Moreno G, Girardello M, Herzog F, Aviron S, Burgess P, **Crous-Duran J**, Ferreiro-Domínguez N, Graves A, Hartel T, Măcicăsan V, Kay S, Pantera A, Varga A, Plieninger T (2016). Cross-site analysis of place-based ecosystem services in multifunctional landscapes Global Environmental Change. 56: 134–147. DOI: 10.1016/j.gloenvcha.2019.04.002

Abstract: Rural development policies in many Organization for Economic Co-operation and Development (OECD) member countries promote sustainable landscape management with the intention of providing multiple ecosystem services (ES). Yet, it remains unclear which ES benefits are perceived in different landscapes and by different people. We present an assessment of ES benefits perceived and mapped by residents (n=2,301) across 13 multifunctional (deep rural to peri-urban) landscapes in Europe. We identify the most intensively perceived ES benefits, their spatial patterns, and the respondent and landscape characteristics that determine ES benefit perception. We find outdoor recreation, aesthetic values and social interactions are the key ES benefits at local scales. Settlement areas are ES benefit hotspots, but many benefits are also related to forests, waters and mosaic landscapes. We find some ES benefits (e.g. culture and heritage values) are spatially clustered, while many others (e.g. aesthetic values) are dispersed. ES benefit perception is linked to people's relationship with and accessibility to a landscape. Our study discusses how a local perspective can contribute to the development of contextualized and socially acceptable policies for sustainable ES management. We also address conceptual confusion in ES framework and present argumentation regarding the links from services to benefits, and from benefits to different types of values.

Keywords: Cultural ecosystem services, Landscape management, Landscape values, Landscape characteristics, PPGIS, Europe

Rolo V, Hartel T, Aviron S, Berg S, **Crous-Duran J**, Franca A, Mirck J, Palma JHN, Pantera A, Paulo JA, Pulido FJ, Seddaiu G, Thenail C, Varga A, Viaud V, Burgess PJ, Moreno G (2020). Challenges and innovations for improving the resilience of European agroforestry systems of high nature and cultural value: a stakeholder perspective. Sustainability Science. DOI: 10.1007/s11625-020-00826-6

Abstract: Traditional agroforestry systems tend to be recognized by sustainability scientists and progressively also by policy makers as flagships for integrating food production, biodiversity conservation and a wide range of tangible and intangible socio-cultural values. Still, several traditional agroforestry systems of Europe are undergoing severe deterioration because the declining economic profitability of many of these systems, while the perspectives of stakeholders across Europe regarding the socio-economic and environmental dimensions of the sustainability of traditional European agroforestry systems is still not understood. In order to fill this gap, we present results of a participatory research performed with ten stakeholder groups (SG) that worked across Europe in the search of solutions to assure the economic and ecological sustainability of High Nature and Cultural Value Agroforestry (HNCV) agroforestry. Stakeholders included both users and beneficiaries of the HNCV agroforestry systems. First, SGs held open discussions (227 participants) to identify major challenges for the long-term sustainability of HNCV agroforestry. Challenges were classified into four categories production, management, socioeconomic and nature conservation. Second, they responded to structured interviews (120 respondents) that explored the positive and negative perception of 45 issues that concern HNCV agroforestry. Third, innovative solutions were scanned by individual and group discussions to address the four categories of challenges. Solutions matched poorly with the challenges identified, and while challenges were at some extent common across countries, solutions to address them were more case specific. The successful implementation of these solutions requires an in-depth understanding of the diversity of socio-cultural and natural contexts of the HNCV agroforestry systems and building bottom-up proposals and collective actions based on this understanding

Keywords: Adaptive policy, European survey, Mosaic-like agriculture, Participative innovation, Regional-based solution, Silvo-pastoral systems.

Bio-physical modelling

Palma JHN, **Crous-Duran J**, Graves AR, García de Jalón S, Upson M, Oliveira TS, Paulo JA, Ferreiro-Dominguez N, Moreno G, Burgess PJ, de Jalón SG, Upson M, Oliveira TS, Paulo JA, Ferreiro-Domínguez N, Moreno G, Burgess PJ (2017). Integrating belowground carbon dynamics into Yield-SAFE, a parameter sparse agroforestry model. Agroforestry Systems. 92: 1047–1057. DOI: 10.1007/s10457-017-0123-4

Abstract: Agroforestry combines perennial woody elements (e.g. trees) with an agricultural understory (e.g. wheat, pasture) which can also potentially be used by a livestock component. In recent decades, modern agroforestry systems have been proposed at European level as land use alternatives for conventional agricultural systems. The potential range of benefits that modern agroforestry systems can provide includes farm product diversification (food and timber), soil and biodiversity conservation and carbon sequestration, both in woody biomass and the soil. Whilst typically these include benefits such as food and timber provision, potentially, there are benefits in the form of carbon sequestration, both in woody biomass and in the soil. Quantifying the effect of agroforestry systems on soil carbon is important because it is one means by which atmospheric carbon can be sequestered in order to reduce global warming. However, experimental systems that can combine the different alternative features of agroforestry systems are difficult to implement and long-term. For this reason, models are needed to explore these alternatives, in order to determine what benefits different combinations of trees and understory might provide in agroforestry systems. This paper describes the integration of the widely used soil carbon model RothC, a model simulating soil organic carbon turnover, into Yield-SAFE, a parameter sparse model to estimate aboveground biomass in agroforestry systems. The improvement of the Yield-SAFE model focused on the estimation of input plant material into soil (i.e. leaf fall and root mortality) while maintaining the original aspiration for a simple conceptualization of agroforestry modeling, but allowing to feed inputs to a soil carbon module based on RothC. Validation simulations show that the combined model gives predictions consistent with observed data for both SOC dynamics and tree leaf fall. Two case study systems are examined: a cork oak system in South Portugal and apoplar system in the UK, in current and future climate.

Keywords: Ecosystem approach, RothC, Climate change, Soil, Leaves, Root, Resilience

Bio-economic modelling

García de Jalón S, Graves A, Moreno G, Palma JHN, **Crous-Duran J**, Kay S, Burgess PJ (2018). Forage-SAFE: a model for assessing the impact of tree cover on wood pasture profitability. Ecological Modelling. 372: 24–32. DOI: 10.1016/j.ecolmodel.2018.01.017

Abstract: Whilst numerous studies have examined the environmental benefits of introducing additional trees within wood pasture systems few studies have assessed the impact on farm profitability. This paper describes a model, called Forage-SAFE, which has been developed to improve understanding of the management and economics of wood pastures. The model simulates the daily balance between food production and the livestock demand for food to estimate annual farm net margins. Parameters in Forage-SAFE such as tree cover density, carrying capacity, and type of livestock can be modified to analyse their interactions on profitability and to identify optimal managerial decisions against a range of criteria. A modelled *dehesa* wood pasture in South-western Spain was used as a case study to demonstrate the applicability of the model. The results for the modelled *dehesa* showed that for a carrying capacity of 0.44 livestock units per hectare the maximum net margin was achieved at a tree cover of around 53% with a mixture of Iberian pigs (28% of the livestock units) and ruminants (72%). The results also showed that the higher the carrying capacity the more profitable the tree cover was. This was accentuated as the proportion of Iberian pigs increased.

Keywords: Wood pasture, Agroforestry, Tree cover, *dehesa*, Model, Profitability

García de Jalón S, Burgess P, Curiel Yuste J, Moreno G, Graves AR, Palma JH, **Crous-Duran J**, Kay S, Chiabai A (2019). Dry deposition of air pollutants on trees at regional scale: A case study in the Basque Country. Agricultural and Forest Meteorology. 278: 107648. - doi: 10.1016/j.agrformet.2019.107648

Abstract: There is increased interest in the role of trees to reduce air pollution and thereby improve human health and well-being. This study determined the removal of air pollutants by dry deposition of trees across the Basque Country and estimated its annual economic value. A model that calculates the hourly dry deposition of NO_2,O_3, SO_2, CO and PM_{10} on trees at a 1 km x 1 km resolution at a regional scale was developed. The calculated mean annual rates of removal of air pollution across various land uses were 12.9 kg O_3 ha^{-1}, 12.7 kg PM_{10} ha^{-1}, 3.0 kg NO_2 ha^{-1}, 0.8 kg SO_2 ha^{-1} and 0.2 kg CO ha^{-1}. The results were then categorised according to land use in order to determine how much each land use category contributed to reducing air pollution and to determine to what extent trees provided pollution reduction benefits to society. Despite not being located in the areas of highest pollutions, coniferous forests, which cover 25% of the land, were calculated to absorb 21% of the air pollution. Compared to other land uses, coniferous forests were particularly effective in removing air pollution because of their high tree cover density and the duration of leaf life-span. The total economic value provided by the trees in reducing these pollutants in terms of health benefits was estimated to be €60 million yr^{-1} which represented around 0.09% of the Gross Domestic Product of the Basque Country in 2016. Whilst most health impacts from air pollution are in urban areas the results indicate that most air pollution is removed in rural areas.

Keywords: Vegetation, Health, Pollutant, Deposition velocity, Land cover

Landscape scale

Kay S, **Crous-Duran J**, Ferreiro-Domínguez N, García de Jalón S, Graves A, Moreno G, Mosquera-Losada MR, Palma JHN, Roces-Díaz JV, Santiago-Freijanes JJ, Szerencsits E, Weibel R, Herzog F (2017). Spatial similarities between European agroforestry systems and ecosystem services at the landscape scale. Agroforestry Systems. DOI: 10.1007/s10457-017-0132-3

Abstract: Agroforestry systems are known to provide ecosystem services which differ in quantity and quality from conventional agricultural practices and could enhance rural landscapes. In this study we compared ecosystem services provision of agroforestry and non-agroforestry landscapes in case studyregions from three European biogeographical regions: Mediterranean (*montado* and *dehesa*), Continental (orchards and wooded pasture) and Atlantic agroforestry systems (chestnut soutos and hedgerows systems). Seven ecosystem service indicators (two provisioning and five regulating services) were mapped, modelled and assessed. Clear variations in amount and provision of ecosystem services were found between different types of agroforestry systems. Nonetheless regulating ecosystems services were improved in all agroforestry landscapes, with reduced nitrate losses, higher carbon sequestration, reduced soil losses, higher functional biodiversity focussed on pollination and greater habitat diversity reflected in a high proportion of semi-natural habitats. The resultsfor provisioning services were inconsistent. While the annual biomass yield and the groundwater recharge rate tended to be higher in agricultural landscapes without agroforestry systems, the total biomass stock was reduced. These broad relationships were observed within and across the case study regions regardless of the agroforestry type or biogeographical region. Overall our study underlines the positive influence of agroforestry systems on the supply of regulating services and their role to enhance landscape structure.

Keywords: Biodiversity, Biomass production, Carbon sequestration, Erosion, Groundwater recharge, Nitrate leaching, Pollination

Kay S, **Crous-Duran J**, García de Jalón S, Graves A, Palma JHN, Roces-Díaz J V., Szerencsits E, Weibel R, Herzog F, Kay S, Silvestre JC, Robert S, Jose HNP, Herzog F (2018). Landscape-scale modelling of agroforestry ecosystems services in Swiss orchards: a methodological approach. Landscape Ecology. 3: 1633–1644. - doi: 10.1007/s10980-018-0691-3

Abstract: *Context* Agroforestry systems in temperate Europe are known to provide both, provisioning and regulating ecosystem services (ES). Yet, it is poorly understood how these systems affect ES provision at alandscape scale in contrast to agricultural practises. *Objectives* This study aimed at developing a novel, spatially explicit model to assess and quantify bundles of provisioning and regulating ES provided by landscapes with and without agroforestry systems and to test the hypothesis that agroforestry landscapes provide higher amounts of regulating ES than landscapes dominated by monocropping. *Methods* Focussing on ES that are relevant for agroforestry and agricultural practices, we selected six provisioning and regulating ES—''biomass production'', ''groundwater recharge'', ''nutrient retention'', ''soil preservation'', ''carbon storage'', ''habitat and gene pool protection''. Algorithms for quantifying these services were identified, tested, adapted, and applied in a traditional cherry orchard landscape in Switzerland, as a case study. Eight landscape test sites of 1 km 9 1 km, four dominated by agroforestry and four dominated by agriculture, were mapped and used as baseline for the model. *Results* We found that the provisioning ES, namely the annual biomass yield, was higher in landscape test sites with agriculture, while the regulating ES were better represented in landscape test sites with agroforestry. The differences were found to be statistically significant for the indicators annual biomass yield, groundwater recharge rate, nitrate leaching, annual carbon sequestration, flowering resources, and share of semi-natural habitats. *Conclusions* This approach provides an example for spatially explicit quantification of provisioning and regulating ES and is suitable for comparing different land use scenarios at landscape scale.

Keywords: Biodiversity, Cherry orchard, Climate change mitigation, Erosion, Landscape water balance, Lonsdorf model, Nitrate leaching

Kay S, Graves AR, Palma JHN, Moreno G, Roces-Díaz JV., Aviron S, Chouvardas D, **Crous-Duran J**, Ferreiro-Domínguez N, García de Jalón S, Măcicăşan V, Mosquera-Losada MR, Pantera A, Santiago-Freijanes JJ, Szerencsits E, Torralba M, Burgess PJ, Herzog F (2019). Agroforestry is paying off – Economic evaluation of ecosystem services in European landscapes with and without agroforestry systems. Ecosystem Services. 36. DOI: 10.1016/j.ecoser.2019.100896

Abstract: The study assessed the economic performance of marketable ecosystem services (ES) (biomass production) and non-marketable ecosystem services and dis-services (groundwater, nutrient loss, soil loss, carbon sequestration, pollination deficit) in 11 contrasting European landscapes dominated by agroforestry land use compared to business as usual agricultural practice. The productivity and profitability of the farming activities and the associated ES were quantified using environmental modelling and economic valuation. After accounting for labour and machinery costs the financial value of the outputs of Mediterranean agroforestry systems tended to be greater than the corresponding agricultural system; but in Atlantic and Continental regions the agricultural system tended to be more profitable. However, when economic values for the associated ES were included, the relative profitability of agroforestry increased. Agroforestry landscapes: (i) were associated to reduced externalities of pollution from nutrient and soil losses, and (ii) generated additional benefits from carbon capture and storage and thus generated an overall higher economic gain. Our findings underline how a market system that includes the values of broader ES would result in land use change favouring multifunctional agroforestry. Imposing penalties for dis-services or payments for services would reflect their real-world prices and would make agroforestry a more financially profitable system.

Keywords: Biomass production, Carbon storage, Soil loss, External cost, Nutrient loss, Pollination deficit

Kay S, Rega C, Moreno G, den Herder M, Palma JHN, Borek R, **Crous-Duran J**, Freese D, Giannitsopoulos M, Graves AR, Jäger M, Lamersdorf N, Memedemin D, Mosquera-Losada R, Pantera A, Paracchini ML, Paris P, Roces-Díaz JV., Rolo V, Rosati A, Sandor M, Smith J, Szerencsits E, Varga A, Viaud V, Wawer R, Burgess PJ, Herzog F (2019). Agroforestry creates carbon sinks whilst enhancing the environment in agricultural landscapes in Europe. Land Use Policy. 83: 581–593. DOI: 10.1016/j.landusepol.2019.02.025

Abstract: Agroforestry, relative to conventional agriculture, contributes significantly to carbon sequestration, increases a range of regulating ecosystem services, and enhances biodiversity. Using a transdisciplinary approach, we combined scientific and technical knowledge to evaluate nine environmental pressures in terms of ecosystem services in European farmland and assessed the carbon storage potential of suitable agroforestry systems, proposed by regional experts. First, regions with potential environmental pressures were identified with respect to soil health (soil erosion by water and wind, low soil organic carbon), water quality (water pollution by nitrates, salinization by irrigation), areas affected by climate change (rising temperature), and by underprovision in biodiversity (pollination and pest control pressures, loss of soil biodiversity). The maps were overlaid to identify areas where several pressures accumulate. In total, 94.4% of farmlands suffer from at least one environmental pressure, pastures being less affected than arable lands. Regional hotspots were located in north-western France, Denmark, Central Spain, north and south-western Italy, Greece, and eastern Romania. The 10% of the area with the highest number of accumulated pressures were defined as Priority Areas, where the implementation of agroforestry could be particularly effective. In a second step, European agroforestry experts were asked to propose agroforestry practices suitable for the Priority Areas they were familiar with, and identified 64 different systems covering a wide range of practices. These ranged from hedgerows on field boundaries to fast growing coppices or scattered single tree systems. Third, for each proposed system, the carbon storage potential was assessed based on data from the literature and the results were scaled-up to the Priority Areas. As expected, given the wide range of agroforestry practices identified, the carbon sequestration potentials ranged between 0.09 and 7.29 t C ha^{-1} a^{-1}. Implementing agroforestry on the Priority

Areas could lead to a sequestration of 2.1 to 63.9 million t C a^{-1} (7.78 and 234.85 million t CO_2eq a^{-1}) depending on the type of agroforestry. This corresponds to between 1.4 and 43.4% of European agricultural greenhouse gas (GHG) emissions. Moreover, promoting agroforestry in the Priority Areas would contribute to mitigate the environmental pressures identified there. We conclude that the strategic and spatially targeted establishment of agroforestry systems could provide an effective means of meeting EU policy objectives on GHG emissions whilst providing a range of other important benefits.

Keywords: Carbon storage, Climate change mitigation, Ecosystem services, Farmland, Resource protection, Spatial deficit analysis

Other publications

den Herder M, Moreno G, Mosquera-Losada RM, Palma JHN, Sidiropoulou A, Santiago Freijanes JJ, **Crous-Duran J**, Paulo JA, Tomé M, Pantera A, Papanastasis VP, Mantzanas K, Pachana P, Papadopoulos A, Plieninger T, Burgess PJ. Current extent and stratification of agroforestry in the European Union. Agriculture, Ecosystems & Environment 241, 121-132. DOI: 10.1016/j.agee.2017.03.005

Abstract: An accurate and objective estimate on the extent of agroforestry in Europe is critical for the development of supporting policies. For this reason, a more harmonised and uniform Pan-European estimate is needed. The aim of this study was to quantify and map the distribution of agroforestry in the European Union. We classified agroforestry into three main types of agroforestry systems: arable agroforestry, livestock agroforestry and high value tree agroforestry. These three classes are partly overlapping as high value tree agroforestry can be part of either arable or livestock agroforestry. Agroforestry areas were mapped using LUCAS Land Use and Land Cover data (Eurostat, 2015). By identifying certain combinations of primary and secondary land cover and/or land management it was possible to identify agroforestry points and stratify them in the three different systems. According to our estimate using the LUCAS database the total area under agroforestry in the EU 27 is about 15.4 million ha which is equivalent to about 3.6% of the territorial area and 8.8% of the utilised agricultural area. Of our three studied systems, livestock agroforestry covers about 15.1 million ha which is by far the largest area. High value tree agroforestry and arable agroforestry cover 1.1 and 0.3 million ha respectively. Spain (5.6 million ha), France (1.6 million ha), Greece (1.6 million ha), Italy (1.4 million ha), Portugal (1.2 million ha), Romania (0.9 million ha) and Bulgaria (0.9 million ha) have the largest absolute area of agroforestry. However the extent of agroforestry, expressed as a proportion of the utilised agricultural area (UAA), is greatest in countries like Cyprus (40% of UAA), Portugal (32% of UAA) and Greece (31% of UAA). A cluster analysis revealed that a high abundance of agroforestry areas can be found in the south-west quadrat of the Iberian Peninsula, the south of France, Sardinia, south and central Italy, central and north-east Greece, south and central Bulgaria, and central Romania. Since the data were collected and analysed in a uniform manner it is now possible to make comparisons between countries and identify regions in Europe

where agroforestry is already widely practiced and areas where there are opportunities for practicing agroforestry on a larger area and introducing novel practices. In addition, with this method it is possible to make more precise estimates on the extent of agroforestry in Europe and changes over time. Because agroforestry covers a considerable part of the agricultural land in the EU, it is crucial that it gets a more prominent and clearer place in EU statistical reporting in order to provide decision makers with more reliable information on the extent and nature of agroforestry. Reliable information, in turn, should help to guide policy development and implementation, and the evaluation of the impact of agricultural and other policies on agroforestry.

Keywords: Land use, Land cover, High natural and cultural value, High value trees, Land use/cover area frame survey (LUCAS)

Gidey T, Oliveira TS, **Crous-Duran J**, Palma JHN. Using the yield-SAFE model to assess the impacts of climate change on yield of coffee (*Coffea arabica* L.) under agroforestry and monoculture systems. Agroforestry Systems. 5. DOI: 10.1007/s10457-019-00369-5

Abstract: Ethiopia economy depends strongly on Coffea arabica production. Coffee, like many other crops, is sensitive to climate change and recent studies have suggested that future changes in climate will have a negative impact on its yield and quality. An urgent development and application of strategies against negative impacts of climate change on coffee production is important. Agroforestry-based system is one of the strategies that may ensure sustainable coffee production amidst likelihood future impacts of climate change. This system involves the combination of trees in buffer extremes thereby modifying microclimate conditions. This paper assessed coffee production under: (1) coffee monoculture and (2) coffee grown using agroforestry system, under: (a) current climate and (b) two different future climate change scenarios. The study focused on two representative coffee growing regions of Ethiopia under different soil, climate and elevation conditions. A process-based growth model (yield-SAFE) was used to simulate coffee production for a time horizon of 40 years. Climate change scenarios considered were: representative concentration pathways (RCP) 4.5 and 8.5. The results revealed that in monoculture systems, the current coffee yields are between 1200 and 1250 kg ha-1 year-1, with expected decrease between 4–38 and 20–60% in scenarios RCP 4.5 and 8.5, respectively. However, in agroforestry systems, the current yields are between 1600 and 2200 kg ha-1 - year-1, the decrease was lower, ranging between 4–13 and 16–25% in RCP 4.5 and 8.5 scenarios, respectively. From the results, it can be concluded that coffeeproduction under agroforestry systems has a higher level of resilience when facing future climate change and reinforce the idea of using this type of management in the near future for adapting climate change negative impacts on coffee production.

Keywords: *Albizia gummifera*, CORDEX, Ethiopia, HADCM3 model, Process-based model, System resilience

Gidey T, Hagosb D, Mehammedseidc H, Solomond N, Oliveira TS, **Crous-Duran J**, Abiyug A, Negussieh A, Palma JHN. Population status of *Boswellia papyrifera* woodland and prioritizing its conservation interventions using multi-criteria decision model in northern Ethiopia. Heliyon 6 (2020). DOI: https://doi.org/10.1016/j.heliyon.2020.e05139

Abstract: The Boswellia papyrifera woodlands provide considerable economic, ecological and socio-cultural benefits in the drylands of Ethiopia. However, their populations are in rapid decline due to different factors, including lack of all stakeholders' involvement in their management and conservation. As a result, the species is now considered as an endangered demanding an urgent conservation intervention to sustain its survival. This study was carried out in Abergele district, northern Ethiopia to quantify current population structure of the species and prioritize its conservation interventions using the Analytical Hierarchy Process (AHP) modelling approach. The species related data were collected from 32 sample plots randomly established in the woodlands of the study area. Data related to the intervention alternatives for the woodlands conservation were also collected from experts, personal experiences and intensive literature review, and then validated using stakeholders' group discussion. Following this, four alternatives: area exclosure (AEA), silvicultural management (SMA), awareness raising (ARA) and development of management plan (DMPA) were considered for prioritization comparison using the AHP techniques. The results showed that the population structure of the species is unstable and characterized by lack of regeneration and small trees (DBH<28 cm) due to combined factors such as overgrazing, over tapping, illegal agricultural expansion and others. The overall priority ranking value of the stakeholders using AHP techniques also indicated the AEA (with overall rank value of 0.288) and ARA (0.280) as the best alternatives, respectively, for the future B. papyrifera woodlands conservation. From the results, it can then be suggested that all relevant stakeholders with their competing interests should be considered and involved during the introduction of the EA and ARA options into the B. papyrifera woodlands conservation in northern Ethiopia.

Keywords: AHP model; area exclosure; conservation alternatives; frankincense; overgrazing; over tapping; regeneration; stakeholders

Chapter 8 |Conclusions

Social perception of agroforestry

Agroforestry is practised on 15.4 million hectares in Europe, about 3.6% of the total territorial area of the European Union (EU) (den Herder et al. 2017) providing multiple benefits to the environment, to society and to local economies (Eichhorn et al. 2006; Plieninger et al. 2015; Fagerholm et al. 2016). However, until land-managers find the motivation and opportunity to change, the area occupied by agroforestry won't increase in Europe (García de Jalón et al. 2017a). In this sense, several studies have already analysed European farmer's perception of agroforestry and all concluded that for them, what would mostly be improved with these practices, would be the environmental aspect of agriculture (Graves et al. 2009, 2017). Farmers are key elements for this change to happen. Assuming their willigness to improve their methods of production towards more sustainble practices, there is a need to know what other barriers these actors are confronted with, because they are not implementing agroforestry practices in Europe despite the financial help offered. In this sense, Chapter 2 explores in 13 different countries in Europe compiling 344 valid respondents and provided evidence on how agoforestry is perveived by these key actors. The study confirmd biodiversity and wildlife habitats were seen as the main positive aspects of agroforestry while increased need of labour, the complexity of work, the management costs and the administrative burden were important negative aspects associated with agroforestry practices. The results in Chapter 2 emphasize that offering solutions on how land-managers can manage agroforestry operations and how new agroforestry systems perform on their farms in financial terms is essential information that will determine their final adoption.

Bio-physical modelling

Computer models are useful tools for estimating bio-physical growth in agroforestry systems allowing analysis of the different components of trees, crops and/or livestock in different soil and climate systems in a more cost-efficient and timely way than field experiments (Vadas et al. 2013; Ford 1999). During the SAFE project (2001-2005), the bio-physical model Yield-SAFE was developed to simulate the dynamics of tree-crop systems (van der Werf et al. 2007) and several environmental indicators were implemented (Palma et al. 2007a). However, these studies were limited to silvo-arable systems. In order to consider other types of agroforestry systems and additional ecosystem services, the model needed to be improved. The model was integrated with new algorithms for determining the crop water use related to vapour pressure deficit; the growth of pasture; the tree leaf fall and fine root mortality for soil carbon dynamics; the livestock carrying capacity of the systems; the tree fruit production and the integration of soil carbon model called RothC (Palma et al. 2017b). These improvements allowed consideration of a fuller range of Provisioning Ecosystem Services (food, materials and energy) provided by silvo-pastoral systems the *montado* and the Swiss cherry orchards, the silvo-arable systems in the UK and the bioenergy systems in Germany. For these systems a key management option, the tree density was analysed in order to understand how the presence of trees determined the ecological performance. In addition, new algorithms for Regulating Ecosystem Services such as for the volume of soil eroded, the nitrate leached, and carbon sequestration allowed the capacity of these systems to provide these three regulating services to be quantified.

In terms of ecological performance Chapter 3 confirms that by adding trees to monoculture arable or pasture systems, in addition to the opportunity of product diversification, the accumulated energy in the systems is increased, indicating a resource-use complementarity between the different components. Although there are some limitations in using energy as a common measure, the approach provided a means of quantifying the production of Provisioning Ecosystem Services and comparison of performance across different tree densities and land use systems in different environmental conditions. In this sense, soil and climatic conditions greatly determine biomass growth and development and the accumulated energy varied depending on the system location, having low values in the drought stressed Portuguese *montado* and the temperature-limited cherry tree-pasture systems of Switzerland, to relatively high values in the temperate environments of the UK and Germany. However, the increase in energy accumulated due to tree presence was not linear, and tree competition for water and solar radiation increased with tree density, reducing the quantity of understory biomass that could be produced, and the energy accumulated per tree.

Regarding Regulating ES, agroforestry systems have been compared to agricultural and forestry alternatives, stating that they provide a land-use solution for additional environmental benefits while maintaining similar levels of productivity. However, scientific research in this area is scare and up to now there has been no assessment of this pattern at a pan-European scale. The study presented in Chapter 4 aimed to better understand this capacity of agroforestry and specifically determine the role of trees in provision of Regulating ES and how an increase in tree density could affect these benefits. To build up consistency through this work, the systems and alternatives analysed were those presented in Chapter 3 and results showed a general improvement on the supply of the Regulating ES associated with the presence of trees in all four systems. At the same time, the research showed that a consistent improvement of Regulating ES can be expected when introducing trees in the farming landscapes in different environmental regions in Europe.

Agroforestry as a Sustainable Intensification practice

The capacity of agroforestry to ensure food production while maintaining the regulating ecosystem services seen in Chapter 3 and 4 supports previous literature that considered agroforestry as a Sustainable Intensification practices (Godfray and Garnett 2014). In this sense and by taking advantage of the new algorithms implemented in the Yield-SAFE model that allow to quantification of Provisioning and Regulating ES, Chapter 5 applies the Carbon Balance Method to a case study in the *montado* in Portugal. This method compares the carbon emissions and the carbon sequestered (balance) derived from the production of a certain quantity of a product in a certain area and allows comparison of production in different systems. The method was applied to the wheat production under crop monoculture and agroforestry systems in Portugal confirming agroforestry as a sustainable intensification method which enables carbon sequestration whilst ensuring the production of wheat.

Bio-economic modelling

The bio-economic modelling exercise undertaken in Chapter 6 combined the outputs of the bio-physical growth model (Yield-SAFE) and a bio-economic model called Farm-SAFE.

The systems analysed were the *dehesa/montado,* in this case in Spain, the silvo-arable systems in the UK and cherry tree orchards in Switzerland. And this bio-economic exercise was considered from two different perspectives: from a farmer prespective by determining the farm profitability or from the societal perspective by including a societal monetary valuation of the main environmental benefits provided (externalities). The study shows that from a financial point-of-view, the most profitable alternatives are always arable options as are able to generate the highest incomes. But these options are also those associated to higher environmental damages. By including monetary valuation of externalities, agroforestry and tree-only systems were shown to be preferable from a societal perspective in the UK. For the Spanish case study tree growth was too slow to compensate for the private benefits from arable agriculture and for Switzerland the large public subsidies made arable and agroforestry options preferable. In this sense, the study demonstrates that location, grants and monetary valuation of externalities can have effect on the final decisions made by land-managers and confirms the need to adapt to the philosophy behind the current Common Agricultural Policy were current levels of subsidies are biased towards no-tree or low-tree density practices, while agroforestry, providing higher Provisioning and Regulating ES and thus a higher societal value than monocropping systems.

Revisiting the objectives

The first objective was related to the productivity off agroforestry practices compared to land-use monoculture alternatives. The use of the bio-physical model (Yield-SAFE), at different European environments considering different edapho-climatic conditions, showed that in general, by including trees in pasture or arable systems, the overall accumulated energy of the system increased and that accumulated energy per tree was reduced as tree density increased.

The second objective of this thesis was related to demonstrate if agroforestry practices were able to offer higher levels of environmental benefits compared to monoculture alternatives. Using a bio-physical modelling approach and considering the same systems analysed in the previous chapter, results demonstrated that the inclusion of trees in arable or pasture land was associated with an increase in the amount of carbon sequestered, and a reduction in soil erosion and nitrate leaching.

The development of the Carbon Balance Method helped to accomplish the third objective and show that, in the Portuguese case analysed, the use of agroforestry, in wheat production could be considered as a Sustainable Intensification practice. Compared to a monoculture, agroforestry production in well-established systems (e.g. after year 50) ensured similar crop yields while trees provide a stable carbon storage.

The fourth objective was to assess the financial and the economic performance of agroforestry compared to its monocropping alternatives. By using a bio-economic model (Farm-SAFE) results showed that considering public grants, agriculture/pure pastures alternatives generated higher incomes and thus were more attractive from a private perspective, even if these systems were also associated with higher levels of environmental damage. But, by including monetary valuation of the environmental externalities, agroforestry and forestry systems increased their attractiveness. However, at the current prices of externalities, this was only sufficient for agroforestry systems to outperform arable systems in the UK. In Spanish conditions, the low growth rate of trees resulted in low levels of carbon sequestration, and in the Swiss case study, the high levels of subsidies available for the monoculture alternatives meant agroforestry was less profitable from a societal perspective. Both the rate of tree growth and grant subsidies are therefore important considerations in the development of public policies and financial support associated with tree integration and agroforestry implementation.

This work attempts to provide evidence on typical performance and profitability questions posed by land managers whilst offering a broader vision of the potential of including trees in farmland as a strategy to ensure food production whilst enhancing environmental benefits. In this sense, evidence is given that agroforestry can offer a good financial performance for farmers from food production whilst also offering good economic returns for society in the form of environmental benefits. Given the climate crisis and the various environmental challenges society faces, this is a win-win solution for both farmers and society. Currently, interests run in opposite directions, and the gap has increased over time. But with this study, it has been shown that the use of agroforestry could allow the private interests of the famers for profit and the interest of the public for environmental benefits could be brought closer together and allowed to move in the same direction.

Main accomplishments

The development of this Thesis lead to the following major accomplishments or findings:

- The upgrade of a process-based model (Yield-SAFE) for the quantification of the Provisioning Ecosystem Services supplied by arable, agroforestry or forestry systems in Europe.
- To implement to this Yield-SAFE model upgraded version additional methodologies for the quantification of three Regulating Ecosystem Services supplied affecting soil (soil erosion), water (nitrate leaching) and air (GHG emissions, carbon sequestration).
- The combination of a process-based model with a Life Cycle Assessment approach for the development of the Carbon Balance Method, an innovative method for the comparison of different food production options and evaluate their potential as Sustainable Intensification practice.
- Offer robust information to land-managers and wider society of the potential productivity of certain agroforestry systems in Europe, the environmental benefits offered and the related financial and economic viability.
- A large potential applicability of the models and methods developed that were tested in four systems in Europe but could be easily adapted for its use in other areas with different edapho-climatic conditions and tree, crop and livestock elements.

Future research

The framework developed consisted in several methods and tools that were used for four selected systems in Europe with specific characteristics of tree, crop and livestock species. However, Europe covers a great may edapho-climatic conditions, crops and trees. This approach could also be useful for assessing the performance of other systems, species, crops, sequential crops or tree/crop/livestock densities to build knowledge on how agroforestry can contribute to agricultural systems in other parts of Europe. Improvements and further research could also:

- Enable the Yield-SAFE model to consider the microclimatic effects associated to the presence of trees and that have a direct effect on the temperature, evaporation, wind speed and livestock growth.
- Include additional methodologies for the estimation of other very specific products with high financial value such as cork in the Portuguese *montado*.
- Apply the Carbon Balance Method developed in Chapter 5 for other systems with the final aim of developing a consistent database of potential sustainable intensification practices.
- Develop a spatial version of the tool to determine the best locations for a selected system.

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Appendix

Appendix 1

Table A1. Tree parameters values used for the Yield-SAFE model for holm oak, cherry tree and poplar for timber and for short rotation coppice.

Parameter	Description	unit	Holm oak (Quercus ilex subsp. rotundifólia)	Cherry tree (*Prunus avium*)	Poplar (*Populus* spp.)	Ppoplar SRC (Populus nigra × Poplar maximowiczii)
Management parameters						
DOYplanting	Day of planting	Julian day	2	2	2	94
DOYpruning	Day of pruning	Julian day	300	350	350	365
Pruning height	Pruning height	m	10	10	10	25
pbiomass	Proportion of biomass removed per prune	0-1	0.1	0.1	0.1	0.99
pshoots	Proportion of shoots removed per prune	0-1	0.05	0.02	0.1	0.2
Maximum proportion of bole	Maximum proportion of bole	1	0.4	0.7	0.5	0.5
Bheight	Maximum bole height	m	4.2	3	8	8.68
DOYthining	Day of thinning	Julian day	305	300	300	365
Site factor			1	1	1	1
Initial conditions						
TreeDensity	number of tree per m2	Tree m-2	-	-	-	-
nShoots0 =	Initial number of shoots	tree-1	0.6	1.8	0.62	1
Biomass0 =	Initial biomass per tree	g tree-1	55	100	100	40
Boleheight0		m	0	1	0	0
LA0 =	Initial leave area	m2 tree-1	0.01	0.5	0	0
Parameters						
ap	Power function to describe relationship between tree height and diameter	unitless	0.5	1	1	1
doybudburst	The day of year when budburst occurs	Julian day	-1	130	100	105

doyleaffall	The day of year when leaves fall. If perennial provide a value higher than 366	Julian day	-	290	300	300
epst	radiation use efficiency	g MJ^{-1}	0.17	0.56	1.40	0.98
F	Form Factor. relates to tree volume, height and diameter	unitless	0.60	0.30	0.37	0.37
gammat	Water use efficiency	m^3 g^{-1}	0.000 46	0.000 20	0.000 40	0.000 20
Kta	Parameter A for radiation extinction coefficient	unitless				
Ktb	Parameter B for radiation extinction coefficient	unitless	14665 75	46475 5.57	16062 65.06	63996 0
Kmain	Fraction of Biomass needed for maintenance respiration	0-1	0.000 1	0.000 08	0.000 1	0.000 1
LA max	Maximum leaf area of a tree	m^{-2}	400	350	500	400
LAsbMax	Maximum leaf area for a single bud	m^{-2}	0.025	0.041 8	0.050 0	0.025 0
NshootsMax	Maximum number of buds on a tree		16000	8367	10000	16000
ratiobranch	Ratio of branches to total biomass	0-1	0.3	0.7	0.30	0.85
ratiotimber	Ratio of timber to total biomass	0-1	0.7	0.3	0.69	0.15
Wood density	Wood density	gm^{-3}	80000 0	60800 0	41000 0	36500 0
pFcritt	Critical pF value for tree, above which tree starts to drought induction	unitless	4.00	4.00	4.00	4.00
PWPt	Permanent wilting point	unitless	4.20	4.20	4.20	4.20
Sigmaheight	Ratio of height to diameter	unitless	120	15	70.31	120
dsigma/density	Response of height/diameter to density	unitless	0.63	0.00	0.00	0.00
Canopywidth/depth	Ratio of maximum width to canopy depth	unitless	1.00	0.50	0.60	0.53
TreeTau	Number of days after bud burst to reach 63.2% of final leaf area	days	10.00	10.00	10.00	10.00
DOYleaffallstart	DOY when leaves no longer grow and start to fall	1-365	105	240	300	292
LeafLeafFallEnd	DOY when leaves no longer fall	1-365	304	330	330	312
fLeafFall	Proportion of leaf area that will fall (1 = deciduous).	0-1	0.30	1	1	1
Weigth single leaf	Weight of a single leaf	g	0.06	0.50	0.50	0.50
Area Single leaf	Area of a single leaf	cm^2	3.00	20.00	84.00	84.00
SLA	Specific leaf area	cm^2 g^{-1}	50.00	170	168	168

CCL	Ratio of carbon content in leaves	0-1	0.25	0.25	0.25	0.25
RSR	Root to shoot ratio (IPCC broadleaves=0.25; conifers=0.2)	0-1	0.10	0.10	0.10	0.10
fFR	Proportion of fine roots from root biomass	0-1	0.50	0.50	0.50	0.50
fCCL	Ratio of carbon content in leaves	0-1	0.50	0.50	0.50	0.50
fCCRt	Ratio of carbon content in tree roots	0-1	0.22	0.22	0.22	0.22
PiSR	Ratio of structural root mass to aboveground biomass	0-1	50000	50000	50000	50000
r	Length of fine roots per unit of structure root	m g^{-1}	0.00	0.00	0.00	0.00
LeafUME	Utilizable Metabolizable Energy from leaves	MJ tDM^{-1}	-	-	-	-
BranchUME	Utilizable Metabolizable Energy from branches	MJ tDM^{-1}	-	-	-	-
FruUME	Utilizable Metabolizable Energy from fruit	MJ tDM^{-1}	17600	7000	-	-
FruitName	fruit name	-	acorn	cherry	-	-
Frup	Fruit productivity per canopy area	g m^2 LAI	40	120	-	-
FruitFallingDays	Number of days when 95% of fruit falls	days	100	100	-	-
FruitDOYPeak	DOY when fruit fall peak occurs		307	210	-	-
FruitWeight	Weight of a single fruit	g $piece^{-1}$	3.50	13.00		
GCV_w	Grooss calorific value of wood	MJ tDM^{-1}	14000	18260	19380	19380
SRC sp?	Is it a short rotation coppice species?	1=yes 0=no	0	0	0	1
reg_prun_freq	Number of years between pruning	years	12	3	-	4
pbiomass_regprun	Proportion of biomass removed per regular pruning	0-1	0.1	0.01	-	0.95
min_th_prun	Minumum tree height for pruning	m	1	3	-	-
Fruit for livestock?	Is the fruit eaten by livestock?	1=yes 0=no	1	0	0	0
Wood for materials?	Is wood used for materials?	1=yes 0=no	0	1	1	0

Appendix 2

Table A2. Crop parameters values used for the Yield-SAFE model.

Parameter	Description	Unit	Wheat (*Triticum durum*)	Barley (*Hordeum vulgare*)	Oilseed (*Brassica napus*)	*Montado* pastures	Swiss pastures	Sugar beet (*Beta vulgaris*)
Management parameters								
DOYsowing	Day of sowing	Julian day	-45	-60	-116	-1	-1	107
DOYharvest	Day of harvest (if threshold not reached)	Julian day	300	300	225	10000	305	260
Override DOYHarvest by calendar (0=Use above rules; 1=Use Calendar)		0/1	0	0	0	0	0	0
To	Temperature threshold	°C	5	5	5	5	5	5
Tsumemerge	Temperature sum to emergence	°Cd	57	57	79	0	160	57
TsumRB	Tsum at which partitioning starts to decline	°Cd	456	456	500	1000	432	456
TsumRE	Tsum at which partitioning to leaves = 0	°Cd	464	464	1300	1100	1030	800
Tsumharvest	Temperature sum to harvest	°Cd	1312	1312	2000	10000	10000	10000
Initial conditions								
BiomassCrop0	Initial biomass	g	10	10	10	10	10	50
Initial leaf area	Initial leaf area	m2 m-2	0.1	0.1	0.1	0.18	0.1	0.075
CropPartition 2leav	Partition to the leaves at emergence	0-1	0.8	0.8	0.8	0.8	0.65	0.5
Parameters								
epsc	Potential growth	g MJ-1	1.85	1.34	1.00	1.70	1.00	0.7
gammac	Water needed to produce 1 gram of crop biomass when VPD=1KPa	m3 g-1	0.00020	0.00025	0.00020	0.00050	0.00040	0.00058
HIcrop1	Harvest index	g g-1	0.6	0.5	0.3	0.8	0.9	0.95
HIcrop2	Harvest index	g g-1	0.3	0.4	0.6	0.2	0.1	0.05

kc	Radiation extinction coefficient		0.7	0.7	0.7	0.7	0.7	0.7
pFcritc	Critical pF value for crop	log(cm)	3.20	2.90	3.20	2.90	3.20	3.2
PWPc	Permanent wilting point for crop	log(cm)	4.2	4.2	4.2	4.2	4.2	4.2
Thetacrop1	Moisture content of the crop (wet basis)		0	0	0	0	0	0
Thetacrop2	Moisture content of the crop (wet basis)		0	0	0	0	0	0
CropSLA	Specific leaf area	m2 g-1	0.005	0.005	0.02	0.0015	0.08	0.08
Site factor		0/1	1	1	1	1	1	1
RSRc	Root-to-shoot ratio - proportion of belowground to above ground biomass	0-1	0.4	0.4	0.4	0.4	0.4	3
fCCRc	Ratio of carbon content in crop roots	0-1	0.3	0.3	0.3	0.1	0.1	0.3
CCAGstraw	Ratio of carbon content in crop straw	0-1	0.5	0.5	0.5	0.5	0.5	0.5
CCAGgrain	Ratio of carbon content in crop grain	0-1	0.5	0.5	0.5	0.5	0.5	0.5
StrawResidue	Above ground residue left afer harvest	0-1	0.1	0.1	0.1	0.1	0z	0.1
CropUME	Utilizable Metabolizable Energy of crop	MJ tDM-1	16630	16690	19450	10270	10500	13500
StrawUME	Utilizable Metabolizable Energy of straw	MJ tDM-1	17300	16100	1400	10270	10500	8000
Crop2Livestock	Use crop harvest to feed livestock	1=yes 0=no	0	0	0	1	1	0
DE	Digestibilty energy (usually 45-55 for low quality forages)	%	50	50	50	50	50	50
Kmainc_m	Maintenance respiration coefficient (fraction of biomass)	g g-1	0.01	0	0.01	0.037	0.03	0.0001
Kmainc_g	Amount of carbon respired to maintain existing biomass	g g-1	0.54	0	0.54	0.54	0.54	0.0001
Pasture/Grass?	Controller for crop manager to pick crop yield	1=yes 0=no	0	0	0	1	1	0
Tuber?	Is the crop a tuber?	1=yes 0=no	0	0	0	0	0	1

Appendix 3

Supplementary data associated with this article can be found, in the online version, at https://figshare.com/s/7438d683d2d80a049631

Appendix 4

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 18**) from the tree and the crop was estimated as:

$$U = \begin{cases} \dfrac{Y_c}{\alpha} + \lambda * \beta_t & if \quad Y_c < \dfrac{Y_{max}}{2} \\ \dfrac{4 * Y_c - Y_{max}}{2 * \alpha} + \lambda * \beta_t & if \quad Y_c \geq \dfrac{Y_{max}}{2} \end{cases} \qquad \text{Equation 18}$$

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 19**).

$$\alpha = \frac{1}{NC_c + NC_r * \frac{Y_r}{Y_c}} \qquad \text{Equation 19}$$

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 (Crous-Duran et al. 2014). Y_r is the residue yield (kg ha^{-1} y^{-1}). The λ coefficient is given as:

$$\lambda = C_{tab} + C_{tbg} RSR \qquad \text{Equation 20}$$

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 5

Table B1. Assumptions for associated costs in the analysis**

Country	Crop	Grain price	Seed rate	Fertiliser rate			Variable costs[a]	Fixed costs[b]	Labour costs
		(€ t^{-1})	(kg ha^{-1})	(kg N ha^{-1})	(kg P_2O_5 ha^{-1})	(kg K_2O ha^{-1})	(€ ha^{-1})	(€ ha^{-1})	(€ 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
ES	Oat	159	100	250	250	250	240	114	27
	Grass	159	45	0	150	0	48	100	22
CH	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 B2. Assumptions for annual governmental support as grants per country and land-use

	Product	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 B3. 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 B4. 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)[-]

Country	Scenario	Option	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

[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

Appendix 6

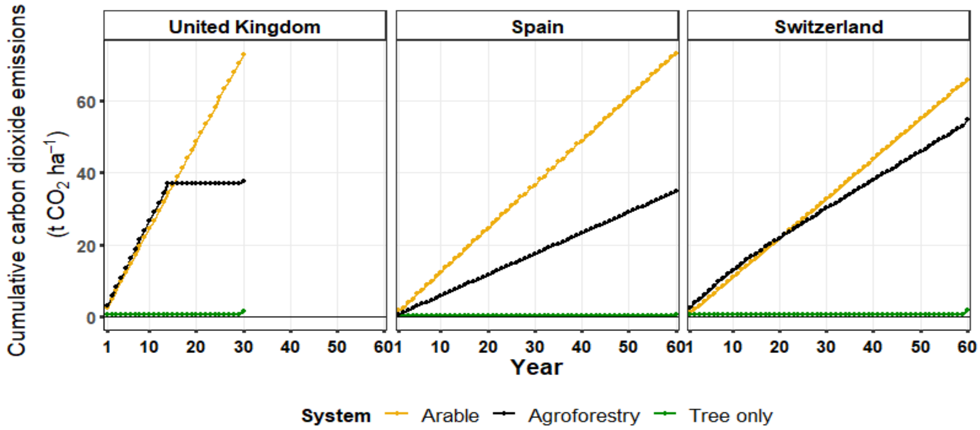

Figure C1. Modelled cumulative CO_2 emissions for the Arable, Agroforestry and Tree-only for UK, Spain and Switzerland over 30, 60 and 60 years respectively

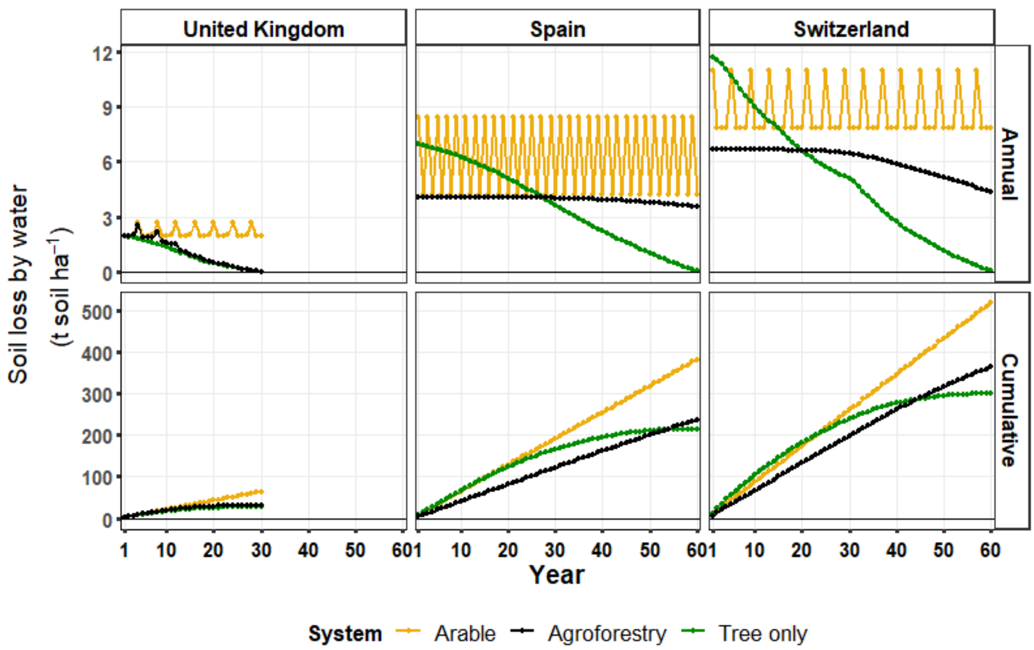

Figure C2. Modelled annual and cumulative soil loss by water (t ha^{-1} y^{-1}) for the Arable, Agroforestry and