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Agroforestry is paying off

- Economic evaluation of ecosystem services in European landscapes with and without agroforestry systems

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Abstract (200 words): 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

1 Introduction

The European agricultural economy relies on revenue from the sale of its agricultural products and thus its success is strongly linked to global prices (Hill and Bradley 2015). The minimum price at which it is profitable to supply these products depends on production costs such as labour, machinery, and fertilisers and other agrochemical inputs. The negative environmental effects or dis-services associated with agricultural production, such as pollution from fertiliser, soil degradation, and biodiversity losses (Zhang et al. 2007), also known as external costs, are not included in the prices paid for agricultural products, and are often experienced by third parties (Tilman et al. 2002; Zander et al. 2016).

During recent decades, the European Common Agricultural Policy (CAP) has provided financial support for agricultural production and rural development (European Commission 2016). Although an increasing share of those payments is linked to environmental performance of farming (pillar II, cross compliance), the effectiveness and efficiency of those financial instruments is regularly questioned (Pe´er et al. 2017). It is therefore anticipated that the next funding period (post 2020) will further strengthen the link between financial support and the improvement of the environment and social well-being, as well as addressing climate change (Council of the European Union 2017).

Agroecological practices, often based on lower agrochemical inputs and higher labour inputs, are increasingly highlighted as promising agricultural systems to reach the goal of environmental and social improvement and favour ecosystem services (ES) (Wezel et al. 2014). ES are defined as the provisioning, regulating and cultural benefits human-beings obtain from ecosystems (MEA 2003; Haines-Young & Potschin 2013). However, these agro-ecological systems are often less profitable than intensive production systems under current subsidy and price schemes and this can hamper their adoption (Ponisio et al. 2014). One example of an agro-ecological multi-functional approach are agroforestry systems. Agroforestry is the incorporation of woody elements on agricultural fields; it simultaneously generates food, fodder, and woody material (Somarriba 1992; European Commission 2013). Moreover, agroforestry can provide ES and multi-environmental functions such as erosion control, reduced nutrient loss, and carbon storage (Torralba et al. 2016) and is thus valued by farmers (García de Jalón et al. 2018a; Rois-Díaz et al. 2018).

Currently, these environmental benefits from agro-ecological approaches that promote ES are typically not monetarized and hence are not included in the market value of the most profitable production system. Palma et al. (2007) integrated monetary and environmental benefits in a multicriteria analysis and concluded that – if they were well designed – agroforestry systems are the preferable land use when environmental benefits are accounted for. In 2010, the Economics of Ecosystems and Biodiversity Report (TEEB 2010) valued services perceived as goods by human beings and distinguished between use and non-use values. According to neoclassical economics the use value was separated into (i) direct use value, (ii) indirect use value and (iii) (quasi) option value. The first two features are premised on market-based cost methods, the last one uses mitigation or non-market cost methods. The ES valuation approach adopted a use value perspective, evolved to a monetary valuation and ended as exchange value or commodity (Costanza et al. 2017). The question remains of how to cash ES in markets (Gómez-Baggethun et al. 2010; Muradian et al. 2010). E.g. Costanza et al. (1997) proposed general values per biome and ecosystem service derived from "willingness-to-pay" studies. The authors already pointed out that the approach is limited for valuing public goods of which people might not be aware such as clean air, clean water, or climate regulation. In recent literature, valuing schemes for ES are divided into payments for ES such as price-based incentives for watershed protection (Bennett et al. 2014) or carbon sequestration (Caparros et al. 2007) and markets for ES e.g. carbon emission trading (Boyce 2018). These payment schemes suffer the problem that e.g. the causal relationship between land use and its service is difficult to define (Muradian et al. 2010) and incomplete information leads to uncertainties and estimations of values (Gómez-Baggethun et al. 2010). However, prices are a tool to value products or services and summarize different ES into one common unit. In the case of carbon markets, prices are also used to regulate emissions (Boyce 2018). Transparent comparisons including both market and non-market values associated with agricultural production are therefore needed for socially beneficial decision-making (e.g. Brenner et al., 2010; Zander et al., 2016).

This study assessed the monetary use values composed of producer surplus and mitigation costs of provisioning and regulating ES for landscapes with and without agroforestry systems. Taking eleven traditional agroforestry landscapes in Europe as an example, we assessed one marketable ES (biomass production) and five non-marketable ES and dis-services (groundwater, nutrient loss, soil loss, carbon sequestration, and pollination deficit) in landscape test sites with and without agroforestry in each region. This research investigated three specific questions: 1) Can sales of marketable ES from agroforestry landscapes match those of landscapes dominated by "business-as-usual" agriculture under current market conditions in different parts of Europe? 2) Do these results change when valuing the (non-market) regulating ES services and dis-services? 3) How sensitive are the results to changes in ES prices?

2 Material and methods

In order to capture the environmental variability and the diversity of agroforestry systems, the study was undertaken in eleven case study regions (> 50 km^2) across the Mediterranean, Continental, and Atlantic regions of Europe. In each case study region, eight landscape test sites (LTS) of 1 km x 1 km were randomly selected, of which four LTS were dominated by agricultural land (NAF, non-agroforestry) and the other four were dominated by agroforestry land (AF). In the NAF LTS the typical agricultural practice of the specific region was analysed and assessed as economic baseline and represents the "business as usual (BAU) alternative". The selection process and further data on each case study region are presented by Moreno et al. (2017).

A total of 88 LTS were assessed, of which 44 NAF LTS provided the economic BAU baseline. In all LTS, the habitats and agroforestry trees were mapped, and ES indicators modelled. In this context the landscape scale represents the aggregation of the four NAF and the four AF LTS, respectively, in a case study region.

2.1 Case study regions

The study regions represent a wide range of agroforestry systems in Europe including scattered wood pastures (e.g. broadleaf-trees in dehesas in Spain or coniferous trees in Switzerland), high value trees systems (e.g. cherry orchards in Switzerland, olives groves in Greece), and wind break systems (e.g. bocage in France or hedgerows in the United Kingdom) as listed in Table 1 and shown in Figure 1.

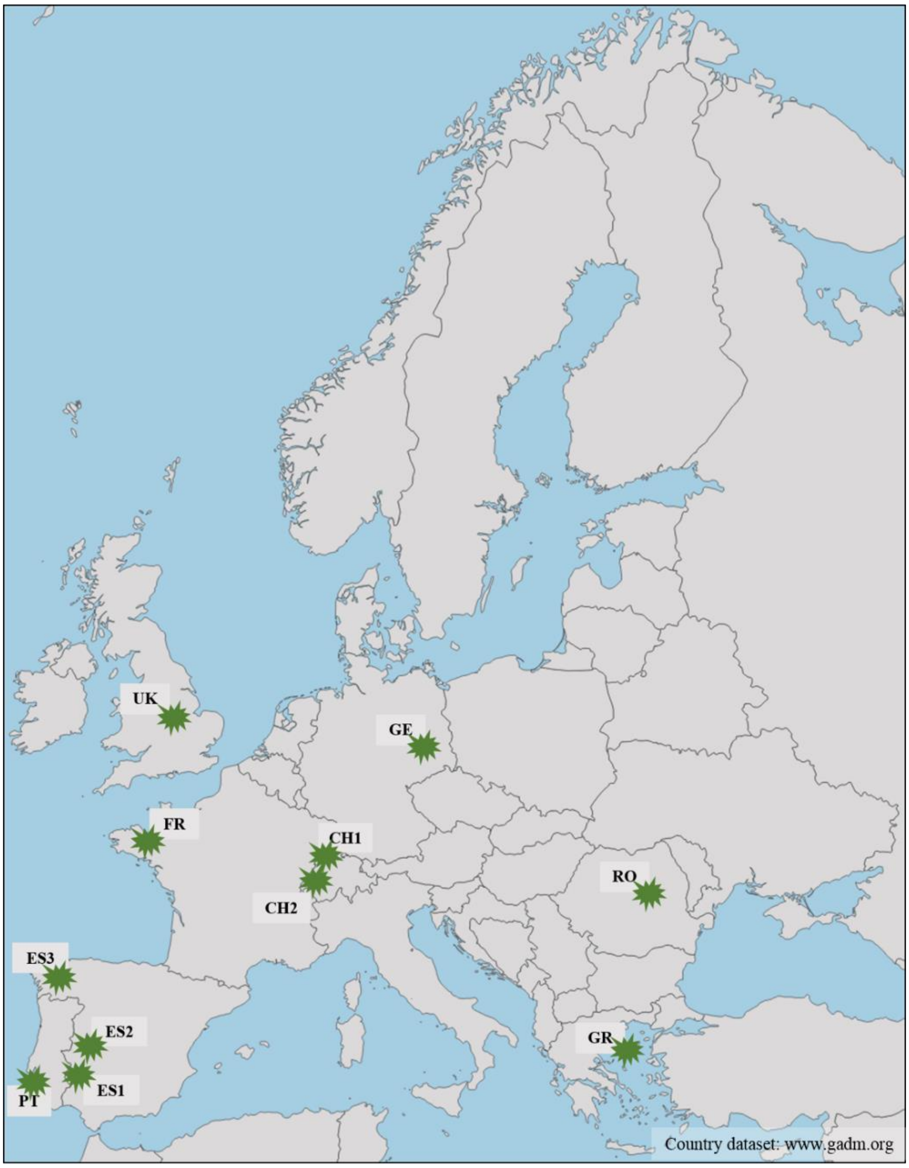

Figure 1: Location of the eleven case study regions.

Table 1: Case study regions and the dominating agricultural (NAF, business as usual) and agroforestry (AF, alternative) system.

Biogeo-graphical region	**Country**	**Abb.**	**Agricultural NAF landscapes (Business as Usual, BAU Baseline)**	**Agroforestry, AF landscapes (Alternative I)**
Mediterranean	Portugal	PT	Open pasture	Montado - Wood pasture (Cork oak, *Quercus suber* L.)
	Greece	GR	Intensive olive groves (*Olea europaea* L.)	Intercrop olive groves (*Olea europaea* L.)
	Spain	ES1	Open pasture	Dehesa - Wood pasture (Holm oak, *Quercus ilex* L.)
	Spain	ES2	Arable farming	Intercrop oak (Holm oak, *Quercus ilex* L.)
Continental	Romania	RO	Open pasture	Wood pasture (Common Oak, *Quercus robur* L.)
	Switzerland	CH1	Open pasture and arable farming	Fruit orchard (Cherry, *Prunus avium* L.)

	Germany	GE	Arable farming	Hedgerow landscape with arable farming (mixed species)
	Switzerland	CH2	Open pasture	Wood pasture (Spruce, *Picea abies* L.)
Atlantic	France	FR	Mixed arable-pasture systems	Bocage - Mixed arable-pasture systems fenced by hedgerows (mixed species)
	Spain	ES3	Open pasture and arable farming	Chestnut soutos (*Castanaea sativa* Miller)
	United Kingdom	UK	Arable farming	Hedgerow landscape with arable farming (mixed species)

2.2 Ecosystem service indicators

One marketable (biomass production) and five non-marketable ES and dis-services (groundwater, nutrient loss, soil loss, carbon sequestration, and pollination deficit) were assessed. The EcoYield-SAFE agroforestry model (Palma et al. 2018) was used to predict biomass production [Unit: t ha^{-1} a^{-1} separately for crop and/or woody material] and aboveground carbon storage [Unit: t C ha^{-1} a^{-1}]. Belowground carbon storage was predicted by YASSO 0.7 (Liski et al. 2005) [Unit: t C ha^{-1} a^{-1}]. The groundwater recharge [Unit: mm ha^{-1} a^{-1}] was based on the general water balance including the evapotranspiration equation by FAO (Allen et al. 1998). Nutrient leaching [Unit: kg N ha^{-1} a^{-1}] was determined by the MODIFFUS 2.0 model (Hürdler et al. 2015), the RUSLE equation (Renard et al. 1997) was used to assess soil loss [Unit: t ha^{-1} a^{-1}], and the pollination service assessment was based on the Lonsdorf equation (Lonsdorf et al. 2009). A spatially explicit model (resolution 2 x 2 m) was used to model these six indicators in 88 LTS (8 LTS x 11 regions) (Kay et al. 2018a, b).

The economic assessment was based on the biophysical evaluation of the six modelled ES indicators. A potential double counting of ES values as highlighted by Fu et al. (2011) was narrowed by using as far as possible independent and static models for each indicator. They were estimated as summarized in the two following sections.

2.3 Valuation and prices of market ecosystem services

<u>Biomass production:</u> The market value of biomass production for food, fodder and woody components was calculated using FAO's compendium "*Producer Prices – Annual per Country*" for each crop (FAO 2017a), the UNECE/FAO TIMBER database "W*ood Prices*" (UNECE/FAO 2017) for timber and the farm accountancy data network (FADN) index *"Total output / Total input (SE132)"* (FADN 2017). The FADN index accounts for the monetary benefit of crop and livestock production and the specific costs. Further information can be found in Appendix I. Overheads are provided on an annual basis for each European country. This was

then used to recalculate the general net profit of crop and timber products by excluding machinery and labour input, which were included in the price datasets. All values are mean values of the years 2010-2014.

The net financial benefit of biomass production per unit weight (Units: € t^{-1}) was determined from the difference between the total output and the total input, which was derived from the total output divided by the FADN index (Eq. 1).

$$B_{Biomass} = T_o - \frac{T_o}{i} \qquad \text{[Equation 1]}$$

$B_{Biomass}$ = Benefits of biomass production per tonne [€ t^{-1}]
To = total output = FAO Producer Prices per crop [€ t^{-1}]
i = FADN index (Farm Accountancy Data Network)

Based on these assumptions, the net financial benefit of biomass production ranged from 0.43 € t^{-1} for wood chips in Switzerland to 802.6 € t^{-1} for walnuts in Greece.

2.4 Valuation and potential prices of non-market ecosystem services and dis-services

Groundwater recharge: Depending on the availability and quality of water resources, the prices per unit indicated in literature varied from 0 to > 4 € m^{-3} depending on the specific country (Roo et al. 2012; JRC Water Portal 2017).

Carbon storage: During recent years, the value of a tonne of carbon dioxide (CO_{2eq}, 3.7C) traded on the European Energy Exchange (EEX) ranged from 2.95 to 8.54 € t^{-1}, with a mean value of about 5 € t^{-1} CO_{2eq} or 18.5 € t^{-1} C (EEX, 2017). Worldwide carbon pricing initiatives use internal prices between 1 and 140 € t^{-1} C (Zechter et al. 2016), the social cost of CO_2 was estimated to range between 5 and 65 \$ t^{-1} (around 5 to 55 € t^{-1} CO_2, Greenstone et al., 2013), and the UN recommend a minimum of 100 \$ t^{-1} C (approximately 85 € t^{-1} C) to maintain global warming within the 1.5 to 2-degree Celsius pathway (United Nations Global Compact 2016).

Nutrient loss: The environmental costs associated with the dis-services of nitrate losses into groundwater are summarized by Brink et al. (2011) and range from 0 to 4 € kg^{-1} N. Recent studies from Denmark and United Kingdom used values of 8 € and 8.4 € kg^{-1} N respectively (OXERA 2006; Jacobsen 2017).

Soil loss: Soil is an important component of agricultural production. Its degradation can lead to a loss of productivity and cause additional off-site (external) costs for compensation and reparation. For the UK, OXERA (2006) used the value of 6.41 € t^{-1} that it costs to remove sediment from domestic water supplies. Schwegler (2014) found that the environmental cost of this dis-service was between 0.9 and 23 € t^{-1}.

Pollination deficits: The dis-service assumed here is assessed in those parts of the LTS where pollination services are deficient. In these areas, crop yield was reduced by the specified requirement for pollination. For example, cherry production is 65% dependent on pollination (Gallai et al. 2009); in pollination deficit areas, the cherry yield was thus assumed to decline up to 65%. For each crop within pollination deficit areas, the biophysical demand for pollination, based on Gallai et al. (2009), was multiplied by the biomass benefit.

2.5 Summary of the net ecosystem service value

Equation 2 describes the benefits and costs associated with the modelled ES. The value (*V*) of the indicator (I) for the benefit or cost of a particular ES is the product of the annual quantity of that indicator (*Q*) multiplied by the monetary value calculated for one unit of that indicator. Table 2 shows the price range and the monetary value (MV_I) of each assessed indicator.

$$V_{\mathrm{I}} = \mathrm{Q}_{\mathrm{I}} * MV_{\mathrm{I}} \qquad \text{[Equation 2]}$$

Table 2: Summary of prices-ranges for ecosystem service indicators and the used monetary values.

Indicators		Unit	Price range	References	Used monetary value (MV_I)
Services	Biomass production	€ t-1	0.43 - 802.6 depending on crop and country	(FADN, 2017; FAO, 2017b; UNECE/FAO, 2017)	0.43 - 802.6 depending on crop and country
	Groundwater recharge	€ m-3	0.0 – 4.0 depending on country	(JRC Water Portal, 2017; Roo et al., 2012)	0.0 – 4.0 depending on country
	Carbon storage	€ t C-1	1.0 – 140.0	(European Energy Exchange (EEX), 2017; Zechter et al., 2016)	5
Dis-Services	Nutrient loss	€ kg N-1	0.0 – 8.4	(García de Jalón et al., 2017b; Jacobsen, 2017; OXERA, 2006)	4
	Soil loss	€ t -1	0.9 – 23.0	(García de Jalón et al., 2017b; Schwegler, 2014)	6.41
	Pollination deficits	€ t-1	0.43 - 802.6 depending on crop and country	(FADN, 2017; FAO, 2017b; Gallai et al., 2009; UNECE/FAO, 2017)	0.43 - 802.6 depending on crop and country

In the final step of the analysis, the services (S) and dis-services (D) were aggregated to provide a net economic value of the combined impact of the ES (NET ES_{value}) by applying Equation 3.

$$NET\ ES_{value} = S_{Biomass} + S_{Water} + S_{Carbon} - D_{Nutrient} - D_{Soil} - D_{Pollination}\ \text{[Equation 3]}$$

with the benefits of biomass production service ($S_{Biomass}$), groundwater (S_{Water}), carbon storage (S_{Carbon}), and the costs for dis-services nutrient loss ($D_{Nutrient}$), soil loss (D_{Soil}) and yield losses caused by reduced pollination ($D_{Pollination}$). The result was expressed for each LTS [Units: € ha^{-1} a^{-1}]. Figure 2 shows an example of the Greek case study region (GR) with four AF (AF1, AF2, etc.) and four NAF LTS (NAF1, NAF2, etc.). The biogeographical comparison was done for the Atlantic, Continental, and Mediterranean regions. Detailed results for each case study region can be found in the Appendix.

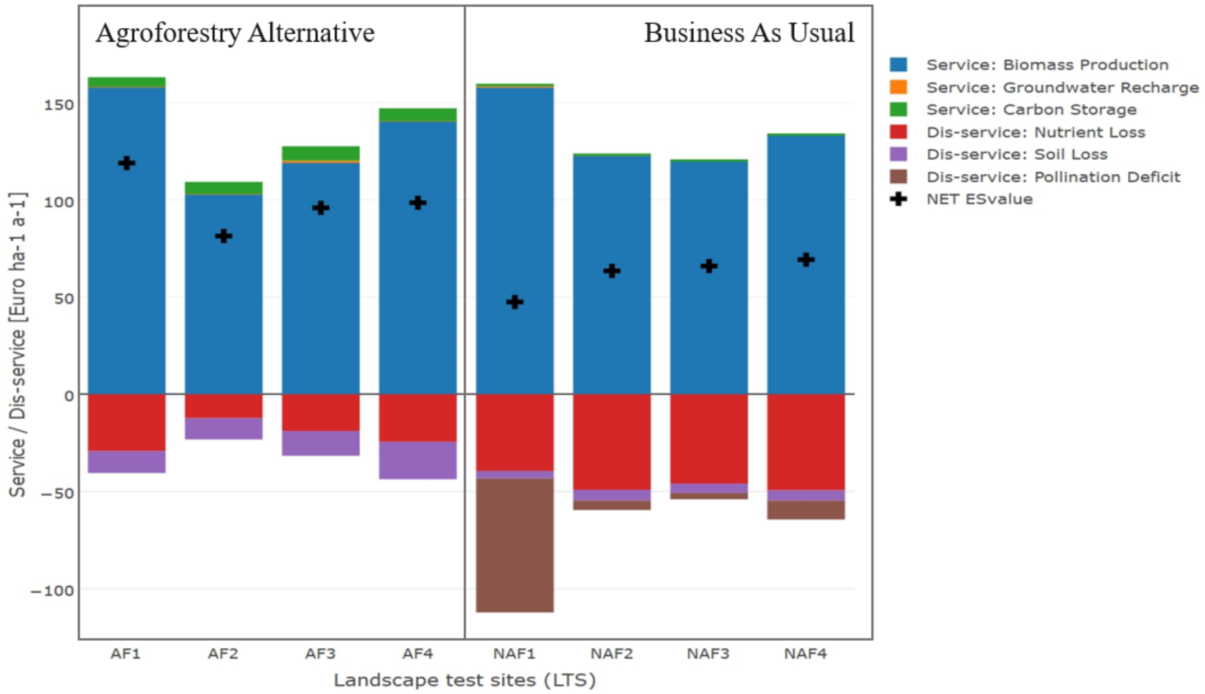

Figure 2: Visualisation of net ecosystem services value (NET ES_{value}) composition including service and dis-service indicators of biomass production, groundwater, carbon storage, nutrient loss, soil loss, and pollination deficit. Indicators were assessed in each landscape test site (LTS) and summarized to NET ES_{valu}.(black cross). The figure shows an example of the Greek case study region with four agroforestry (AF1, AF2, etc.) and four non-agroforestry LTS (NAF1, NAF2, etc.) as Business-As-Usual baseline.

2.6 Evaluation of threshold prices

In order to identify a threshold cost for each pollutant where the benefit of the non-agroforestry landscape (NAF) matched the agroforestry (AF) landscape for nutrient emissions, soil losses, and carbon storage, we conducted a detailed analysis of the range of prices found in literature. The intersection points - where landscapes with and without agroforestry systems (AF vs NAF LTS) are on equal economic terms - were determined.

Nutrient loss expressed as nitrate pollution costs were examined in the range between 0 and 8 € kg^{-1} N, soil degradation costs were examined from 0 to 20 € t^{-1} soil and carbon prices were assessed in a range between 0 and 100 € t^{-1} C.

The analyses were conducted using R (R Development Core Team 2016). The figures were created with the R packages ggplot2 (Wickham et al. 2016) and plotly (Sievert et al. 2016) and the maps with QGIS (QGIS Development Team 2015).

3 Results

3.1 Valuation of ecosystem services

3.1.1 Net benefit from biomass production

The mean value for the annual net financial benefit of biomass production (crop and timber products) tended to be higher in agricultural NAF landscapes. On average across all study regions, the mean profit was 36 € ha^{-1} a^{-1} in the NAF landscapes as compared to 29 € ha^{-1} a^{-1} in the AF landscapes (Figure 3). Large differences were found among the biogeographical regions. The oak and olive systems of the Mediterranean landscapes had a mean financial net benefit of 76 € ha^{-1} a^{-1}, and the AF landscapes provided a greater financial revenue from biomass than the NAF landscapes. Atlantic and Continental landscapes were less lucrative, and NAF LTS generated slightly greater financial net benefits than the AF landscapes (Figure 3).

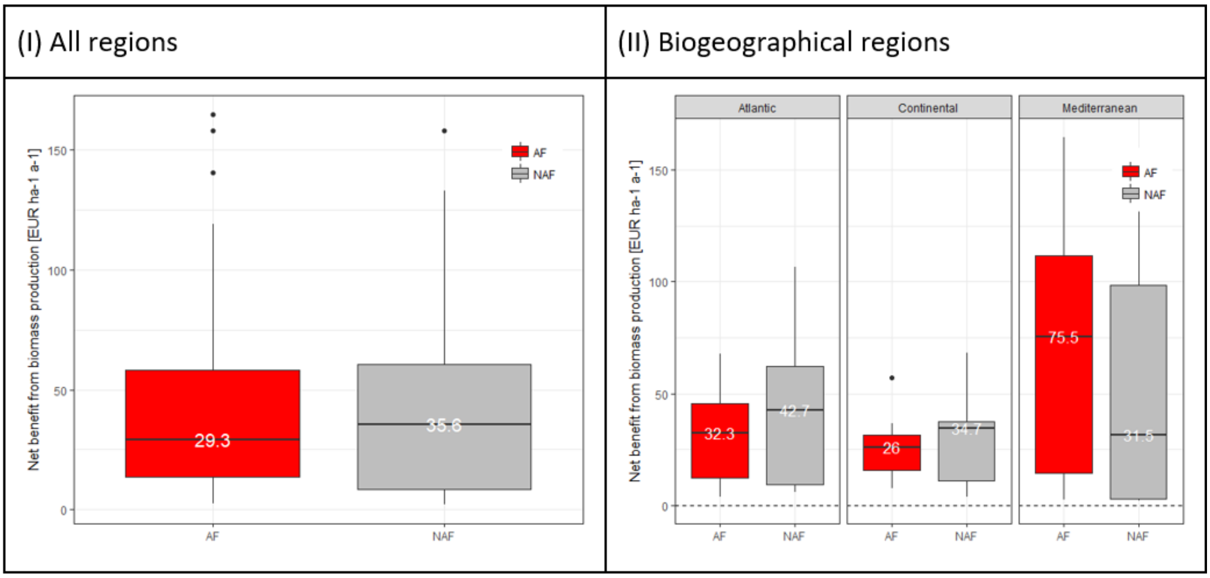

Figure 3: Average net financial benefit of biomass production [€ ha^{-1} a^{-1}] of all 11 cases study regions (I) and divided into biogeographical regions (II) based on landscape test sites [LTS] grouped by land cover categories into agroforestry (AF) and non-agroforestry (NAF, Business as Usual) sites.

3.1.2 Monetary valuation of individual ecosystem services and dis-services

In terms of benefits, the market value of biomass production was greater than the monetary values assigned to groundwater, carbon storage, nutrient and soil losses, and pollination service deficits across all the LTS, reaching as much as 160 € ha^{-1} a^{-1} in some cases (Figure 4). The financial benefit of groundwater recharge was typically less than 2 € ha^{-1} a^{-1}. Carbon sequestration benefits ranged between 15 and 30 € ha^{-1} a^{-1}. In terms of costs, nutrient pollution in water caused costs as great as 150 € ha^{-1} a^{-1} and soil loss costs ranged between 15 and 30 € ha^{-1} a^{-1}. The market value of reduced pollination service was typically minimal across the LTS.

Figure 4 also illustrates the relative performance in monetary terms depending on the proportion of agroforestry in a LTS. Whilst the dis-service nutrient loss was higher in LTS without agroforestry, only a slight difference appeared in the case of the market value of biomass production. The highest values, both positive and negative, occurred in LTS without agroforestry.

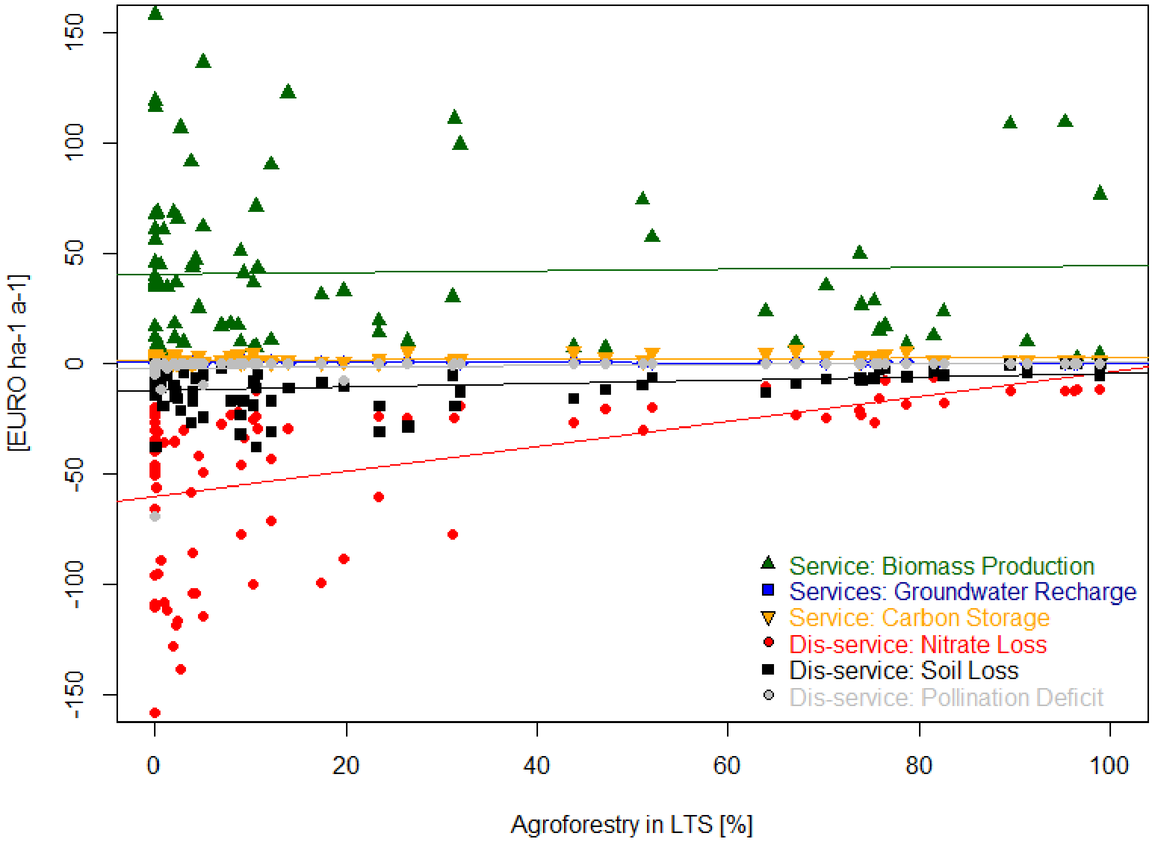

Figure 4: Monetary values [€ ha^{-1} a^{-1}] of ES indicators, depending on the percentage of agroforestry in the landscape test sites (LTS). The coloured lines are the regression line of the measurements.

3.1.3 Integrated assessment of monetary valuation of all ecosystem services and dis-services

The net value of the ES for each LTS was also summed up for the case study regions (Figure 5). The net value of the AF landscapes tended to be greater in all three biogeographical regions, indicating that they provided greater economic welfare to society in comparison to the NAF landscapes. However, in nearly all regions, the net societal values of both, the agricultural and the agroforestry landscapes, were calculated to be negative when externalities were included in the economic analysis. The only exception were the Mediterranean agroforestry landscapes. The highest negative values were found in agricultural landscapes in the Atlantic regions. All results per ES and case study regions are listed in Appendix II.

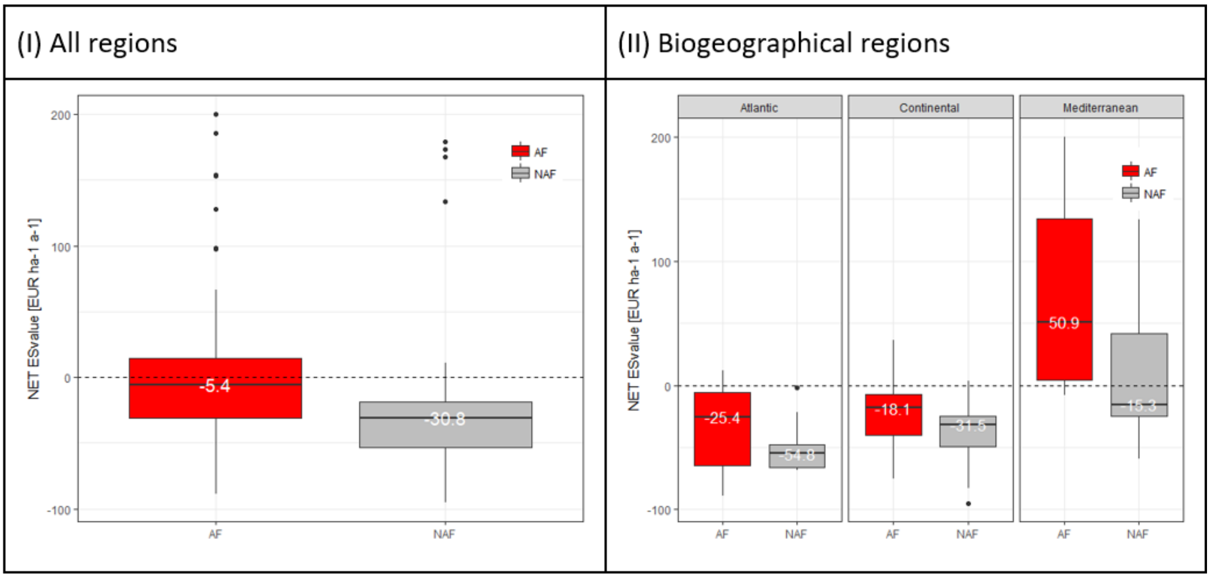

Figure 5: Net ecosystem service value in € ha^{-1} a^{-1} of all 11 cases study regions (I) and divided into biogeographical regions (II) based on landscape test sites [LTS] grouped according to dominating land cover categories into agroforestry (AF) and non-agroforestry (NAF, Business as Usual) LTS.

3.2 Threshold prices

Building on the previous results, threshold values were calculated to identify the ES price level that would be needed for AF systems to become as profitable as the NAF systems (Figure 6). This was done for nutrient loss, soil loss, and carbon storage using the revenue from biomass production to provide a baseline for the NAF LTS, whilst the external costs or benefits for each ES were added individually to the baseline of the AF LTS. In this analysis, cost and prices of the other ES were not accounted for.

3.2.1 Nutrient loss

Figure 6a shows how the economic performance (€ ha^{-1} a^{-1}) in the biogeographic regions decreased as the cost of nutrient losses (€ kg^{-1} N) increased. As the economic output of biomass was used as baseline it remained unchanged. The AF landscapes generally showed a slower decrease in overall profitability as costs of nutrient losses increased, which indicates an overall greater robustness of these systems. The NAF landscapes showed negative economic outcomes at a nutrient emission cost of 3 € kg^{-1} N, whereas AF LTS provided positive returns up to a nutrient emission cost of approximately 5 € kg^{-1} N.

These results differed in the three biogeographical regions. Whilst the AF LTS in the Atlantic and Continental systems were slightly less profitable than NAF when nutrient emission costs were 0 € kg^{-1} N, AF and NAF were equally profitable when the nutrient emission cost was 2.5 € kg^{-1} N. This shows that even though economic output of biomass production is generally lower in Atlantic and Continental AF (Atlantic: AF 32.3 € ha^{-1} a^{-1}, NAF 42.7 € ha^{-1} a^{-1}; Continental: AF 26.0 € ha^{-1} a^{-1}, NAF 34.7 € ha^{-1} a^{-1}), introducing even fairly low costings for nutrient emission would reverse the relationship due to lower nitrate losses in the AF areas. In all three regions, the relative benefit of AF systems increased as the cost of nutrient emission increased.

3.2.2 Soil loss

The soil loss assessment (Figure 6b) showed similar results to the nutrient emission assessment. In general, a rise in the cost of soil erosion resulted in declining economic performance of both AF and NAF relative to the economic output of the biomass only scenario. Again, the economic performance of the AF landscapes suggested greater robustness as decreases in economic performance were less than for NAF as the cost of soil losses increased. While in Atlantic and Continental regions, economic performance of AF was lower at low soil loss costs compared to NAF, the economic performance of AF benefitted from rising costs of soil loss relative to NAF. At values for soil loss of 12 € t^{-1} soil (Continental biogeographic region) and 17 € t^{-1} soil (Atlantic biogeographic region), AF and NAF landscapes produced the same economic outcome. Rising the cost for soil loss by another 5-10 € made all landscapes (AF, NAF) unprofitable in those two regions, whilst in the Mediterranean region, both landscape types remained profitable, at least within the price range investigated.

3.2.3 Carbon sequestration

The results for carbon sequestration (Figure 6c) showed that increasing the value of stored carbon resulted in increases in the economic performance of both AF and NAF systems across all the biogeographic regions. However, the patterns were comparable to the results for nutrient emissions and soil loss. Generally, AF was more profitable than NAF even at modest carbon prices. In Atlantic and Continental biogeographic regions particularly, AF profited from an increasing carbon value and exceeded the economic performance of NAF at most carbon values (thresholds were at approximately 10 € t^{-1} C in the Continental biogeographic regions and 30 € t^{-1} C in the Atlantic biogeographic region; the Mediterranean AF was more profitable at all carbon values).

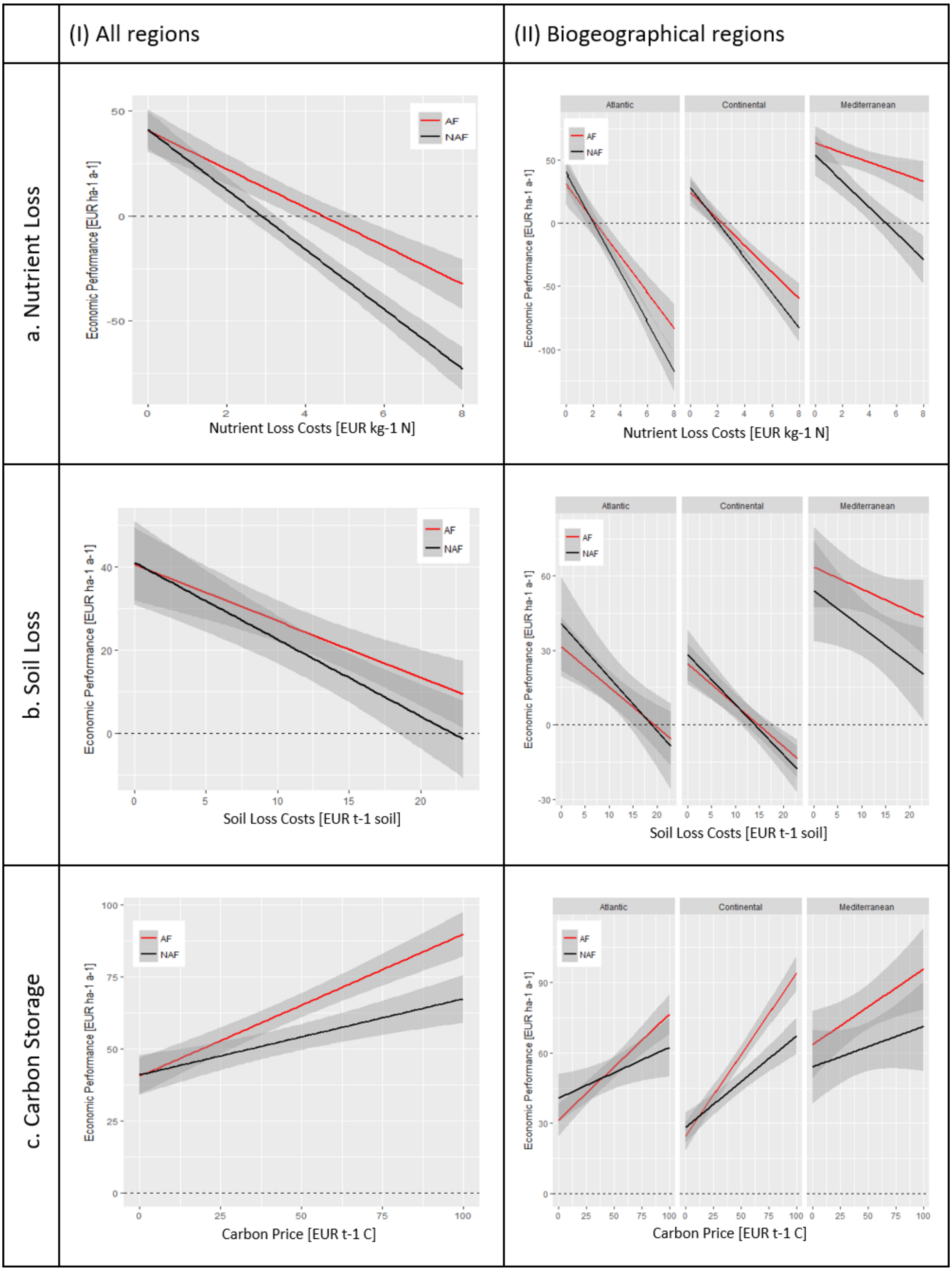

Figure 6: Economic performance of agroforestry (AF) and non-agroforestry (NAF, Business as Usual) for different ecosystem services (a) nutrient emission costs, (b) soil loss costs and (c) carbon prices together with the current sales revenues of biomass production in € ha^{-1} a^{-1} (I) over all 11 cases study regions and (II) divided into biogeographical regions based on landscape test sites [LTS] grouped by dominating land cover categories into AF and NAF LTS.

4 Discussion

This research investigated three questions: 1) How does the societal value of agroforestry landscapes compare with landscapes dominated by agriculture in different parts of Europe if only the values of the products are considered? 2) Do these results change if the values of selected regulating services are included? and 3) How sensitive are the results to changes in ES prices?

The trends we identified are in line with former findings on ES and mitigation provided by agroforestry systems (e.g. Jose, 2009; Moreno et al., 2017a; Tsonkova et al., 2012). Biophysical and economic differences between agroforestry and non-agroforestry practices are clearer at the plot scale (Graves et al., 2007; Palma et al., 2007; Sereke et al., 2015). Our investigation related to the landscape scale because some ES such as soil conservation or pollination services involve spatial interactions that cannot be evaluated at the plot scale. Yet, as we investigated mixed landscapes there were some agroforestry trees even in NAF LTS and vice-versa, which somehow "blurred" the differences between the landscape test sites. Also, the proportion of the land use categories agroforestry, agriculture, forest and others differed from region to region, which led to the high variability observed.

In response to the first research question, in Atlantic and Continental regions of Europe, the market values of the products from agroforestry landscapes were calculated to be generally lower than for non-agroforestry systems. The opposite was observed in Mediterranean regions, where the market value of the products from agroforestry landscapes were calculated to be higher than for the non-agroforestry cases. This was mainly due to the multiple tree products (olives or acorns in addition to timber) and the use of the FADN index in the calculation of the market value of the biomass production, which was between 1.28 and 1.31 in Portugal, Spain and Greece; while in northern and central European countries values around 1.0 were obtained. The agroforestry olive groves in our Greek case study region were already fully productive and therefore profitable (producer price: ~2000 € t^{-1}; net benefit: ~470 € t^{-1}; yield: 100 kg $tree^{-1}$; 1 t ha^{-1} a^{-1} olives, 0.2 t ha^{-1} a^{-1} olive oil; European Commission, 2012; FAO, 2017b; Pantera et al., 2016). According to the European Commission (2012) (intensive) olive production is one of the most important and profitable agricultural activities in southern marginal regions. Whilst olive groves produce in average 2.5 t ha^{-1}, agroforestry production is around 1 t ha^{-1}. After five to seven years, olive systems start to become fully productive and after around year 20 the initial costs are covered and they obtain revenues (Stillitano et al. 2016). This resulted in AF landscapes to have higher sales revenues in Mediterranean regions than NAF landscapes. The multiple gains of dehesas are reflected in their land prices for lease or sale. While open pastures in Spain cost around 5’000 € ha^{-1} and are leased for 53.50 € ha^{-1}, dehesas are on sale for ~8’000 € ha^{-1} and leased for 78.70 € ha^{-1} (Consejería de agricultura 2014; FEDEHESA 2017). This positive economic performance for AF relative to NAF is also reflected in the spread and extent of agroforestry in Mediterranean regions. Den Herder et al. (2017) identified the current extent of AF in Europe and found that the largest areas were in Spain (5.6 million ha), Greece (1.6 million ha), France (1.6 million ha), Italy (1.4 million ha) and Portugal (1.2 million ha).

For AF in Continental and Atlantic regions the situation is different. Sereke et al. (2015), Nerlich et al. (2013) and Eichhorn et al. (2006) have stated that many traditional agroforestry systems are in decline. Highlighting and valuing their environmental role was related to the second research question. Actually, the decision of managing the land as an agroforestry system is not only related to financial profitability but also to other criteria such as to increase the diversification of products, improve biodiversity, animal health and welfare as described by García de Jalón et al. (2017a), Rois-Díaz et al. (2017), and Sereke et al. (2016). This indicates that (some) farmers value ES even if they don’t provide financial benefit. At the policy level, the European environmental (e.g. Water Framework Directive) and agricultural policies (CAP with greening and cross compliance) focus was on the impact of environmental pollution, notably nutrient emissions and soil losses. Here, even small monetary benefits associated with reduced nutrient and soil losses, and – in addition – modest carbon sequestration payments favoured the economic performance of the assessed systems in favour of agroforestry. These findings are echoed by Zander et al. (2016) in their evaluation of the performance of grain legumes, and La Notte et al. (2017) in their evaluation of in-stream nitrogen and reflect the failure of markets to pass costs back to polluters.

The third research question focused on the sensitivity of the outcomes to price changes. Unexpectedly, the value of nutrient emissions was the most important factor affecting the economic performance of the assessed systems, since small changes in prices charged for nutrient losses led to relatively large changes in economic performance. Compared to this, soil losses were of lesser importance, as also observed by García de Jalón et al. (2017b). Even though water pollution by nitrates is addressed by several environmental regulations (e.g. Nitrate Directive, Water Framework Directive), European water prices for irrigation or domestic purposes are surprisingly low. In comparison with the costs and prices assigned to other ES indicators, they thus had a negligible impact on economic performance.

The decline in pollinators and its possible consequence on pollination service has been a key issue at European scale (Zulian et al. 2013; Breeze et al. 2014). However, as enough nesting and foraging resource for wild pollinators were available in all case study landscapes (Kay et al. 2018b), the cost of potentially reduced pollination services had no impact.

Regarding the European climate policy (e.g. EU 2030 Climate and Energy Framework), carbon storage and emission reduction are the most important ES. Agroforestry has the potential to store carbon on agricultural land (Zomer et al. 2016). The United Nations Global Compact (2016) proposes the use of a carbon value of \$100 t^{-1} (approximately 85 € t^{-1} C). If such high carbon prices could be obtained by farmers, this would drastically change the economic performance of many land use systems. Even with a carbon price of 30 € t^{-1} C, landscapes with AF were more profitable compared to NAF LTS.

5 Conclusion

In many parts of Europe, agroforestry systems such as wood pastures and hedgerows remain under threat either due to land abandonment or an increase in mechanization and decline in labour availability. In this study, AF landscapes in Atlantic and Continental regions showed slightly lower market outputs than agricultural areas if the focus was only on marketable provisioning ecosystem services. However, in Mediterranean regions, the marketable outputs from the considered agroforestry systems were typically greater than the associated agricultural system.

When the societal values of regulating ES and dis-services were also accounted for, the aggregated landscape profitability of AF was generally higher than NAF in each region. This was driven by a reduction in societal costs related to lower nutrient and soil losses, and the societal benefits of carbon sequestration. Overall, our study underlined that relatively low costs per ES unit (nutrient emission: > 2.5 € kg^{-1} N; soil loss: > 17 € t^{-1} soil; carbon sequestration > 30 € t^{-1} C) would be sufficient to render AF profitable, at least to match NAF profitability.

Our results show that there is a critical gap in economic assessments that fails to account for ecological and social benefits. This issue needs to be imperatively addressed if international agreements (e.g. European Commission, 2011; UNFCCC, 2015; United Nations, 1992) should have any effect. New methods of accounting for externalities e.g. payments for ecosystem services or other incentives to stimulate farmers and land users to turn towards more socially beneficial forms of land use should be strengthened.

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