Assessment of Ecosystem Services provided by Agroforestry Systems at the Landscape Scale

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Content

Summary

Agriculture, as one of the main land users and key drivers of landscape changes, faces multiple challenges nowadays and in the near future. The need to satisfy the rising demand for high quality food and material is accompanied by the requirement to adapt to a changing climate and to mitigate emissions and pollutions. However, the performance of agricultural land is not singularly related to its production function, but also the need to meet the demands of human-well-being for environmental, regulatory, and social benefits. These multiple and partly-conflicting objectives converge at the landscape level. Landscapes allow for reconciling between the different objectives to enhance overall efficiency and reduce trade-offs. As such, future multifunctional landscapes are expected to be highly productive, sustainable, environmentally friendly and climate-smart.

Agroforestry, a "land use system in which trees are grown in combination with agriculture on the same land", might play a key role in future farming in Europe. Adding trees to agricultural land improves micro-climatic conditions, soil water-holding capacity, habitat diversity and carbon storage while simultaneously producing food, fodder and timber. All of this provides ecosystems services (ES) for farmers and society.

Against this background, the present thesis investigates three main research questions: (1) Does the provision of ecosystem services differ in landscapes with agroforestry compared to landscapes dominated by agriculture? (2) Is this ecosystem service provision related to economic and environmental benefits within these landscapes? and (3) Can agroforestry systems significantly contribute to European climate targets of zero-emission agriculture?

In order to answer the abovementioned questions, an adapted quantitative and transdisciplinary approach at the landscape scale, with in-depth analysis of landscape test sites (LTS) was used. In contrasting landscapes dominated by (a) agroforestry or (b) agriculture, eight LTS of 1 by 1 km spatial resolution were selected and mapped in the field. Bio-economic and environmental modelling were used to quantify seven provisioning and regulating ES (biomass production, groundwater recharge, nutrient retention, soil preservation, carbon sequestration, pollination and habitat diversity) to characterise the performance of the agroforestry and of the agricultural LTS. The outcomes revealed a higher supply of regulating ES in landscapes with agroforestry systems, while provisioning ES were better represented in agricultural landscapes. The same relationship was obtained by applying the spatial model to 12 European agroforestry landscapes

(montado in Portugal, dehesa and soutus in Spain, olive groves in Greece, orchards in Switzerland, bocage in France, hedgerow landscapes in the UK and Germany, and wooded pastures in Romania, Switzerland and Sweden). Traditional agroforestry systems, regardless of type, region and composition, had a beneficial impact on regulating ES at the landscape scale. Nitrate and soil losses were reduced and carbon sequestration, pollination services and the proportion of semi-natural habitats were higher in agroforestry landscapes. Agricultural landscapes were linked to a higher annual biomass yield and a higher groundwater recharge rate.

In the follow-up phase of the study, the economic performance of marketable ES and non-marketable ES in these contrasting landscapes were assessed. The findings showed that agroforestry areas had slightly lower market outputs than agricultural areas if the focus was only on marketable ES. However, when monetary values for non-marketable ES were included, the relative profitability of agroforestry landscapes increased. A gap in economic assessments that fails to account for ecological benefits was detected.

Finally, European priority areas for introducing agroforestry systems were identified by assessing environmental farmland deficits in soil, water, climate, and biodiversity. For each priority area, agroforestry candidates were proposed by regional experts. Systems were highly variable and their ability to capture carbon was evaluated; this evaluation resulted in a storage potential of between 0.09 to 7.29 t C ha^{-1} a^{-1}. Assuming that the priority area, which makes up 8.9% of European farmland, were to be converted to agroforestry, carbon emissions of agriculture could be reduced by up to 43.4 %.

With this in mind, a spatially-explicit model was developed and validated to quantify the ecosystem services supply from agroforestry systems from a landscape perspective; this model could also be used to evaluate different land use scenarios. The modelled outcomes demonstrated that agroforestry had a beneficial impact on landscapes and the potential to mitigate the challenges of climate change while securing food and fodder production, improving the environment and natural resources, and enhancing biodiversity.

Zusammenfassung

Die Landwirtschaft, als grösster Flächennutzer und prägendes Element des Landschaftswandels, steht in naher Zukunft vor grossen Herausforderungen. Eine steigende Nachfrage nach qualitativ-hochwertigen Lebensmitteln und Biomaterialien geht einher mit der Forderung zur Reduktion der Emissionen und Umweltbelastungen und den Anforderungen sich an den Klimawandel anzupassen. Landwirtschaftlich genutzte Flächen sind nicht mehr nur Ressourcenlieferant, ihre Regelungs- und Umweltfunktionen sowie ihr (Nah)Erholungswert treten immer mehr in den Vordergrund. All diese zahlreichen teilweise widerstrebenden Landnutzungsinteressen vereinen sich auf der Landschaftsebene. Diese übergeordnete Managementebene erlaubt es unterschiedlichste Zielsetzungen synergetisch zu vereinen und widerstrebende Tendenzen weitgehendst zu reduzieren. Die multifunktionale Landschaft der Zukunft ist hoch produktiv, nachhaltig, umwelt- und klimafreundlich.

Agroforstwirtschaft, *ein Landnutzungssystem, in welchem Bäume und Landwirtschaft auf dem gleichen Feld kombiniert werden*, könnte in der Agrarproduktion der Zukunft eine entscheidende Rolle in Europa spielen. Bekanntermassen verbessern Bäume in der Agrarlandschaft das Mikroklima, erhöhen die Bodenwasserhaltekapazität, die Habitat-Vielfalt und speichern Kohlenstoff. Gleichzeitig liefern sie Holz, Lebens- und Futtermittel. Sie stellen eine Vielzahl an Ökosystemdienstleistungen (ÖDL) für Landwirte und Gesellschaft bereit.

Aus diesem Kontext heraus ergaben sich drei Forschungsfragen: (1) Welchen Einfluss haben Agroforstsysteme auf die Bereitstellung von ÖDL auf Landschaftsebene im Vergleich zur landwirtschaftlichen Nutzung. (2) Ist die Bereitstellung von ÖDL verbunden mit ökonomischen und ökologischen Vorteilen innerhalb dieser Landschaften? Und (3) können Agroforstsysteme einen wesentlichen Beitrag zu den europäischen Klimazielen bzgl. einer Null-Emission Landwirtschaft beitragen?

Die vorliegende Arbeit folgte einem quantitativen transdisziplinären Ansatz, der sich auf die Landschaftsebene und darin speziell auf Landschafts-Test-Quadrate (LTQ; jeweils 1 x 1 km) konzentrierte. Ausgewählt wurden acht LTQ in kontrastierenden Landschaften mit (a) agroforstwirtschaftlicher oder (b) landwirtschaftlicher Nutzung, die im Feld hinsichtlich ihrer Habitat-Ausstattung kartiert wurden. Im Anschluss wurden Modellierungsansätze genutzt, um sieben ÖDL Indikatoren («Biomasse-Produktion», «Grundwasser-Neubildungsrate», «Nährstoff-Rückhaltung», «Boden-Sicherung», «Kohlenstoff-Sequestrierung», «Schutz von

Habitaten und Genpool») zu quantifizieren. Die Darstellung potenzieller Unterschiede zwischen agroforstlicher und landwirtschaftlicher Nutzung stand bei der Auswahl der ÖDL im Vordergrund. Die Ergebnisse zeigten eine deutlich höhere Bereitstellung von regulierenden ÖDL in Agroforst-Landschaften, während bereitstellende ÖDL in landwirtschaftliche Landschaften dominierten. Die Anwendung des Modells auf weitere 12 europäische Agroforst-Landschaften (französische bocage, griechische Olivienhaine, britische und deutsche Hecken-Landschaften, portugiesische montado; rumänische und schwedische Baumweiden, Schweizer Hochstammwiesen und Wytweiden, sowie spanische dehesa und soutus) bestätigte die gefundenen Zusammenhänge. Traditionelle Agroforstsystems unabhängig des Typs, der Region und der Zusammensetzung hatten einen positiven Einfluss auf die Bereitstellung von regulierenden ÖDL auf Landschaftsebene. Nitrat- und Bodenverluste waren reduziert, Kohlenstoffspeicherung, Bestäuberleistung und der Anteil an halbnatürlichen Habitaten war höher in Agroforst-Landschaften. Landwirtschaftliche Landschaften verzeichneten einen größeren jährlichen Biomasseertrag und eine höhere Grundwasserneubildungsrate.

Auf diesen Ergebnissen aufbauend, wurde die Gesamtwirtschaftsleistung in den kontrastierenden Landschaften auf Basis der vermarktbaren und unvermarktbaren ÖDL bewertet. Es zeigte sich, dass Agroforst-Landschaften im Vergleich zu landwirtschaflich genutzten Landschaften eine geringfügig reduzierte wirtschaftliche Ausbeute generierten. Sobald jedoch die bisher nicht marktfähigen ÖDL in die Berechnung einbezogen wurden, erhöhte sich die relative Rentabilität der Agroforst-Landschaften.

Darüber hinaus wurden europäische Vorrang-Gebiete zur Etablierung von Agroforstsystemen auf Basis von Defizitregionen bzgl. der Boden- und Wasserqualität, der Klimaauswirkungen und Biodiversität-Ausstattung identifiziert. 64 Agroforstsysteme wurden für die Vorrang-Gebiete von regionalen Experten empfohlen. Das Kohlenstoffspeicherpotential der genannten Systeme belief sich auf 0.09 to 7.29 t C ha^{-1} a^{-1}. Unter der Annahme, dass die Vorrang-Gebiete und damit in etwa 8.9% der europäischen Landwirtschaftsfläche, vollständig in eine agroforstwirtschaftliche Bewirtschaftung umgewandelt würden, könnten bis zu 43.4% der heutigen Treibhausgasemissionen der europäischen Landwirtschaft reduziert werden.

Insgesamt unterstreichen die gefundenen Untersuchungsergebnisse, dass Agroforstwirtschaft einen positiven Einfluss auf die Landschaftsebene hat, das Potential bietet den Herausforderungen des Klimawandels zu begegnen und eine beständige, nachhaltige, klimafreundliche Agrarproduktion in Europa zu sichern.

Chapter 1

Introduction

1.1 Motivation

Landscapes – a source of goods, provider of services and determinant of humans' (place) identity – are constantly progressing and changing (Biasi et al., 2016; Plieninger et al., 2016). They alter in composition and spatial configuration as a result of natural processes and human activities (Baessler and Klotz, 2006; Burkhard et al., 2009; Tilley, 2006).

By highlighting the key global drivers which influence landscape changes, international policy essentially identified two main global trends: first, the rising demand on land productivity along with biodiversity losses, and second the changing climate conditions with potential adaptation and mitigation opportunities (United Nations, 2000, 1992). In addition, landscape changes are varying in regional expression. Whereas northern Europe is affected by land intensification, land abandonment is the most important driver in the southern parts (Plieninger et al., 2016).

An in-depth look at global trends revealed:

I: The persistent population growth results in rising demands for agricultural production and (until now) in an unremitting biodiversity loss.

The constantly growing global population (1950: 2.5 billion, 2000: 6.1 billion, 2050: 9.7 billion, United Nations, 2017) goes along with an increased demand for food, fodder and material. The follow-on intensification of agricultural production has resulted in environmental problems such as air and water pollution and a general loss of biodiversity (Koellner and Scholz, 2008; Tilman, 1999). In Europe there exist serious challenges, namely nitrate pollution of water bodies (Van Grinsven et al., 2012), soil health (Tsiafouli et al., 2015) and habitat changes brought about by loss, fragmentation or degradation (EEA, 2018).

Global policy, especially the Sustainable Development Goals as part of the 2030 Agenda of sustainable Development (United Nations, 2015), the Millennium Ecosystem Assessment (MEA, 2003) and the Convention on Biological Diversity and their targets (COP, 2010; UNEP, 2002), aim to increase the productivity of agricultural production, while simultaneously ensuring sustainable development, climate mitigation and adaptation, and the maintenance of biodiversity and ecosystem service flows. These developments were echoed in European regulations such as the Strategic Plan for Biodiversity 2011-2020 in 2010 (COM(2011) 244), the Water Framework Directive (Directive 2000/60/EC) in 2000 and the Soil Thematic Strategy in 2006 (COM(2006)231).

Recommendations of the Food and Agriculture Organization of the United Nations highlight the huge potential of diversified agricultural practices (FAO, 2011) and integrated production systems - such as agroforestry (FAO, 2017a), the integration of trees on agricultural land (Somarriba, 1992) - as very promising. Building mosaic agricultural landscapes can be beneficial for biodiversity (species and ecosystems) while generating higher productivity at the same time. Forms, effects and changes towards the "sustainable intensification" of agricultural production have been discussed by, e.g., MacFadyen et al. (2012), Petersen and Snapp (2015) and Garibaldi et al. (2017). In addition, the concept of Climate-Smart Agriculture (CSA) was launched in 2010, aiming to (i) increase sustainable agricultural productivity, (ii) adapt climate change resilient farming and (iii) reduce greenhouse gas emissions (FAO, 2017a).

II: The global climate change, a result of rising greenhouse gas emissions, increases natural hazards, extreme weather events and has a far-reaching impact on earth, ecosystems and humans.

For several years the World Economic Forum (2018) has ranked the failures of climate change mitigation and adaptation as well as the management of extreme weather events among the top 10 Global Risks. Climate change has and will have an increasing effect on (natural) production, territories and ecosystem services (Blanke et al., 2017; Olesen et al., 2012).

As early as 1994, the UN Framework Convention on Climate Change (UNFCCC, United Nations, 1992) entered into force with the aim of limiting greenhouse gas (GHG) concentrations so as to reach a level which would offer sustainable living conditions for humans and ecosystems. The resulting Kyoto Protocol (UNFCCC, 1998) contained binding emission reduction targets (reduction of 5% GHG in industrialised countries compared to the level of 1990) and management mechanisms (e.g. Emissions Trading). In Paris the 21st Conference of the Parties (COP21 Paris Agreement, UNFCCC, 2015) agreed to bring the global temperature rise below 2 degrees by 2100 and demanded Nationally Determined Contributions from its members. Agriculture, as one of the main sectors of GHG emissions (~11 % of global emissions, FAO 2016), can contribute by storing carbon in soils and biomass. Zomer et al. (2016) showed the important effect of trees on agricultural land for global carbon storage. Accordingly, many developed countries proposed to prioritise agroforestry to contribute to their long-term climate goals (World Agroforestry Centre, 2017). In Europe, the EU proposed the Effort Sharing Regulation, including a "no-debit rule" for agricultural practices. This means "carbon neutrality" by an equal amount of GHG emissions and sequestration. Said regulation is in negotiation (European Parliament, 2017).

In conclusion, future agricultural production will ideally address these main global targets. It should increase agricultural productivity while simultaneously ensuring the maintenance of biodiversity and capturing carbon to contribute to global climate mitigation and adaptation objectives. Additionally, as farming always relies on land and space, the local growing and market conditions, as well as the regional stakeholders, need to be involved. Sustainable production systems (García-Feced et al., 2015; Wezel et al., 2014) using a more holistic landscape view (FAO, 2017a; Scherr et al., 2012) are needed. Agroforestry is seen as one opportunity to address many of these targets (Hart et al., 2017; Jose, 2009).

1.2 Landscape analysis

The sustainable management of multiple goals and various demands on land requires the understanding of landscapes, their patterns and their processes (Jones et al., 2013). Moreover, Hein et al. (2006) showed that various goals, stakeholder interests and environmental services refer to different spatial scales. He distinguished between the ecological and the institutional scale – the place of service generation versus the place of its benefit and management. In addition, Hunziker et al. (2007) and Kienast et al. (2015) investigated humans place attachment and place making and highlighted the differences between space and place. These more holistic approaches lead to the concept of a multipurpose landscape (Minang et al., 2014), also known as “climate-smart landscape” (Scherr et al., 2012). Herein, the landscape is described by referring to its functional interactions, negotiated spaces and multiple scales (Minang et al., 2014).

1.2.1 Definition of landscapes and motivation for landscape analysis

The landscape scale plays an inherent role in politics and management, as it is a geographical limitation of territories and a basis for regulation and governance. According to the European Landscape Convention (Council of Europe, 2000) *"Landscape" means an area, as perceived by people, whose character is the result of the action and interaction of natural and/or human factors; […],* and *"Landscape management" means action, from a perspective of sustainable development, to ensure the regular upkeep of a landscape, so as to guide and harmonise changes which are brought about by social, economic and environmental processes […] (Article 1).* The UNEP Report on how to improve the sustainable use of biodiversity from a landscape perspective (United Nations Environment Programme, 2011) added in *II. 9. […] It is a spatial scale which is important in terms of a continuous flow of key ecosystem services. [...].* Herein, the United Nations highlighted the value of landscapes as a level for planning

framework and management tasks, as a multiple land-use provider, and as a platform on which to combine ecological, socioeconomic and institutional values.

Englund et al. (2017) recently defined landscapes as "*An area viewed at a scale determined by ecological, cultural-historical, social and/or economic considerations*".

1.2.2 From pattern and processes to multifunctional landscapes

Historically, landscape research has been primarily linked to landscape ecology, which was initiated by Carl Troll in the 1930s. The first research boom has been experienced in the late 80s, as Turner (1989) indicated the influence of landscape pattern on ecological functions and processes. She pointed out that the (spatial) connectivity between habitats or their fragmentation influences the quality of ecological functions and ecosystems. A quantification of landscape structures into landscape metrics and indices would offer the potential to quantify and monitor ecological processes and biodiversity just based on maps or orthophotos on different scales.

This was the starting point for various studies which evaluate the relationship between landscape pattern und ecological processes, identifying the essential components of landscapes and developing meaningful landscape metrics (e.g. Baker and Cai 1992; McGarial and Marks 1995; Lausch and Herzog 2002; Turner 2005; Cushman et al. 2008). Even though appropriate indicators were detected, no uniform metric was set, which overall performed satisfying results. Depending on the region and the investigated process the outcomes differed; no single best answer was found (Sayer et al., 2013).

Landscape analysis was essentially based on the configuration and composition of landscape elements (Lausch et al., 2015) and assumed a more diverse heterogenetic mosaic as a proxy for greater biodiversity (Leopold, 1933). Herein, composition characterises the number, the proportion, or the diversity of habitats, while the configuration represents the spatial relationship between landscape elements and their connectivity or complexity. The first was described as landscape pattern, while the second was a proxy for landscape functions and processes (Farina, 2000). Both were quantified and assessed based on spatial-explicit maps. These maps varied depending on the considered landscape unit (natural/landcover classes, Cushman et al. 2008), administrative boundaries (Kienast et al., 2009), format (continuous/raster or discrete/vector), scale and thematic resolution or focus of interest (Bailey et al., 2007b; Lausch and Herzog, 2002).

However, there is still debate surrounding this promising way of easily analysing landscapes and drawing conclusions regarding the inventory and quality of biodiversity and its functions.

The essential limitations of this quantitative analysis are that: (i) maps are models of reality, mostly phenomena- or application-driven and show a human perception-centred classification (Cushman and Huettmann, 2010; Lausch et al., 2015; Schulp and Alkemade, 2011), (ii) the indicators are very grain, extent and spatial scale sensitive (this means that results depend on patch size, boundary and data format) (Bailey et al., 2007b; Mitchell et al., 2015), (iii) not all indicators are valid for all spatial scales (spatially explicit at patch or landscape level) (Dale and Polasky, 2007; Verhagen et al., 2016), (iv) the interpretation of the indicator-function-relationship is difficult and needs to be adequate (Bailey et al., 2010; Kienast et al., 2009) and (v) most maps present a certain period in time, while landscape processes are per se dynamic (Istanbulluoglu and Bras, 2005; Verburg et al., 2013). Users of landscape metrics for valuing landscape functions should be aware of these simplifications and assumptions.

Around 2000 the two-dimensional "pattern-process relationship" was enlarged by further landscape dimensions such as (i) design (Nassauer, 2012), (ii) human place attachment and place making (Kienast et al., 2015; Wartmann and Purves, 2018), and (iii) ecosystem services (Bürgi et al., 2015; Syrbe and Walz, 2012). Termorshuizen and Opdam (2009) proposed to expand these approaches by adding a valuation component, and calculating landscape services. More specifically, the link between landscape pattern and the provision of ecosystem services became important (Syrbe and Walz 2012; Englund et al. 2017, Chapter 1.3). Related questions included whether multifunctional landscapes also provide multiple services (Mitchell et al., 2015) and how landscape pattern influences ecosystem service provision (Jones et al., 2013).

Regarding the discussion concerning adaptation and mitigation of climate change, the concept of "climate-smart landscapes" was formulated (Minang et al., 2014; Scherr et al., 2012). The approach focused on creating synergies between environmental, social and economic functions while reducing trade-offs (Duguma et al., 2014). Exceeding the so-far-descriptive landscape assessment, the approach interacted with multiple stakeholders, asked for their motivation and land management practices, involved the governance and finance sector, and tracked changes (Scherr et al., 2012).

1.3 Ecosystem Services

1.3.1 Definition and classification

The concept of "Ecosystem Services", hereinafter referred to as ES, is a political framework which became popular in 2003 as a result of the United Nations Millennium Ecosystem Assessment (MEA).

While landscape analysis had focused on ecology, processes and biodiversity, the motivation of the ES framework was to value ecosystems for human well-being. ES were defined as *"the benefits people obtain from ecosystems"*. MEA (2003) showed how these ES were degrading on a global scale and provided policy recommendations.

DEFINITION	**Ecosystem services (ES)** are *"benefits people obtain from ecosystems"*. They were grouped into (a) Supporting Services: nutrient cycling, soil formation and primary production (b) Provisioning Services: food, fresh water, wood, fibre and fuel production (c) Regulating Services: climate, flood, disease regulation and water purification (d) Cultural Services: aesthetic, spiritual, educational and recreational values (MEA, 2003).

Years before this framework, the linkage between ecosystems, their functions and human well-being had already been addressed by several research studies (Costanza et al., 1997; Daily, 1997; De Groot, 1994; De Groot et al., 2002; Ehrlich and Ehrlich, 1981; Gómez-Baggethun et al., 2010; Westman, 1977). Discussion focused specifically on the awareness that biodiversity loss and pollution have an effect on ecosystem functions and an impact on society. The term "ecosystem services" was first coined by Ehrlich and Ehrlich (1981). The first definition was provided by Daily (1997), who described ES as "*conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life"*. She had already linked the concept to a preliminary list of 13 indispensable services. Based on this adapted list, Costanza et al. (1997) attempted a first quantification of the monetary value of global natural capital and ecosystem services. The alarming results for the world's economy entered the political arena and contributed to the initiation of the Millennium Ecosystem Assessment (Costanza et al., 2017).

Over the years the term "ecosystem services (concept)" has been used in a particularly dynamic way. Gómez-Baggethun et al. (2010) investigated its development from a description of natural complexity to increased public awareness to an instrument in (financial) markets and a reference value for environment-economic accounting systems. As a result of this turbulent development, the term ES has not yet been consistently defined.

A first classification system of ecosystem functions was presented by De Groot (1992); moreover, a first list of ecosystem services, subdivided into 13 indicators, was provided by Daily (1997). Costanza et al. (1997) built on this list and adapted it to 17 indicators; De Groot et al. (2002) then extended it to a list consisting of 23 indicators.

The political process, started by the MEA in 2003, also promoted classification and quantification approaches. The MEA presented a classification system, in which it distinguished between supporting, provisioning, regulating and cultural ES categories. Ten years later Haines-Young and Potschin (2013) presented the Common International Classification of Ecosystem Services (CICES). It was developed for the System of Environmental and Economic Accounting (SEEA), which was led by the United Nations Statistical Division (UNSD).

CICES is linked to the European "mapping and assessment of ecosystems and their services" (MAES) process (Haines-Young, 2016), which in turn is linked to the European Union's Biodiversity 2020 Strategy (European Commission, 2011; Maes et al., 2016). However, at present, the CICES is mainly a classification system for ES. Standardised methods for the assessment and quantification of ES are still being developed.

CLASSIFICATION

Common International Classification of Ecosystem Services (CICES)

Classification system including around 50 indicators in three categories – Provisioning, Regulating and Maintenance, and Cultural Services.

Table 1: Classification of ES according to CICES

Section	Division	Group
Provisioning	Nutrition	Biomass
		Water
	Materials	Biomass, Fibre
		Water
	Energy	Biomass-based energy sources
		Mechanical energy
Regulation & Maintenance	Mediation of waste, toxics, and other nuisances	Mediation by biota
		Mediation by ecosystems
	Mediation of flows	Mass flows
		Liquid flows
		Gaseous / air flows
	Maintenance of physical, chemical, biological conditions	Lifecycle maintenance, habitat and gene pool protection
		Pest and disease control
		Soil formation and composition
		Water conditions
		Atmospheric composition and climate regulation
Cultural	Physical and experiential interactions with ecosystems and land-/seascapes	Physical and experiential interactions
		Intellectual and representative interactions
	Spiritual, symbolic, and other interactions with ecosystems and land-/seascapes	Spiritual and /or emblematic
		Other cultural outputs

(Haines-Young and Potschin, 2013)

Finally, this gave birth to definitions (e.g. Boyd and Banzhaf 2007; Braat and de Groot 2012), map (e.g. Burkhard et al. 2012; Clec'h et al. 2016) and assessments of ES (e.g. Syswerda & Robertson 2014; Maes et al. 2016). Recent studies have focused on the spatial allocation and quantification of ES (Burkhard et al., 2013) along with assessments of synergies and trade-offs between ES (e.g. Turner et al. 2014; Mouchet et al. 2017); indeed, all of this has been combined to produce the effects of spatial pattern on bundles of ES (Raudsepp-Hearne et al., 2010). The important role of the scale and the connection of ES to landscape pattern has been discussed by, e.g., Hein et al. (2006), Dale and Polasky (2007), Anderson et al. (2009), and Verhagen et al. (2016). Lastly, numerous models have been developed to assess single or multiple ES in various regions, on numerous scales, and with different thematic focuses (InVEST by Nelson

et al. 2009; Co$ting Nature by Faleiros et al.; ARIES by Villa et al. 2014, etc.). Conceptual modelling of multiple ES at the landscape scale is shown by Helfenstein and Kienast (2014).

1.3.2 Ecosystem service valorisation

In 2010 the economic valuation of ES was emphasised. The TEEB Foundation - The Economics of Ecosystems and Biodiversity, an international initiative hosted by the United Nations Environment Programme (UNEP), quantified benefits provided by ES and the cost of ES losses. Based on the "cascade model" from Haines-Young and Potschin (2010), trade-offs between benefits from landscape and the pressure on landscape were identified and monetised. Herein, ES were defined as "*the direct and indirect contributions of ecosystems to human well-being"* (TEEB, 2010).

TEEB (2010) valued services perceived as goods by human beings and distinguished between the preference-based approaches, which assess use and non-use values, and the biophysical approaches, which focus on resilience values or physical costs. According the neoclassical economics, used in the preference approach, 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-base cost methods, the last one uses mitigation or non-market cost methods.

The ES valuation approach transformed from a use value perspective to a monetary value towards an exchange value or commodity. It ended in the question of how to cash ES in markets (Gómez-Baggethun et al., 2010; Muradian et al., 2010). In recent literature valuing schemes for ES are divided into Payments for ES (PES) such as price-based incentives for watershed protection (Bennett et al., 2014) or carbon sequestration (Caparrós et al., 2007) and Markets for ES e.g. carbon emission trading (Boyce, 2018). These payments schemas suffer the problem that e.g. the causal relationship between land use and its service is difficult to define (Muradian et al., 2010) and that these incomplete information lead to estimations of values (Gómez-Baggethun et al., 2010). However, prices are a tool to value productions or services and summarize different ES into one common unit. In the case of carbon, prices are also used to regulate emissions (Boyce, 2018)."

The ES cascade model as presented in Figure 1 is an excellent example of how the ES framework and the different levels of landscape analysis and approaches (Chapter 1.2) work together. While landscapes as space (referred to by Hein et al. 2006 as "ecological scale"; basis of landscape ecology) provide services, the landscape as a place, living and regulating area (also

known as "institutional scale") is the place where human-beings receive benefits (Hunziker et al., 2007). In this frame, ES are the tools used to assess and the unit used to valorise the benefits.

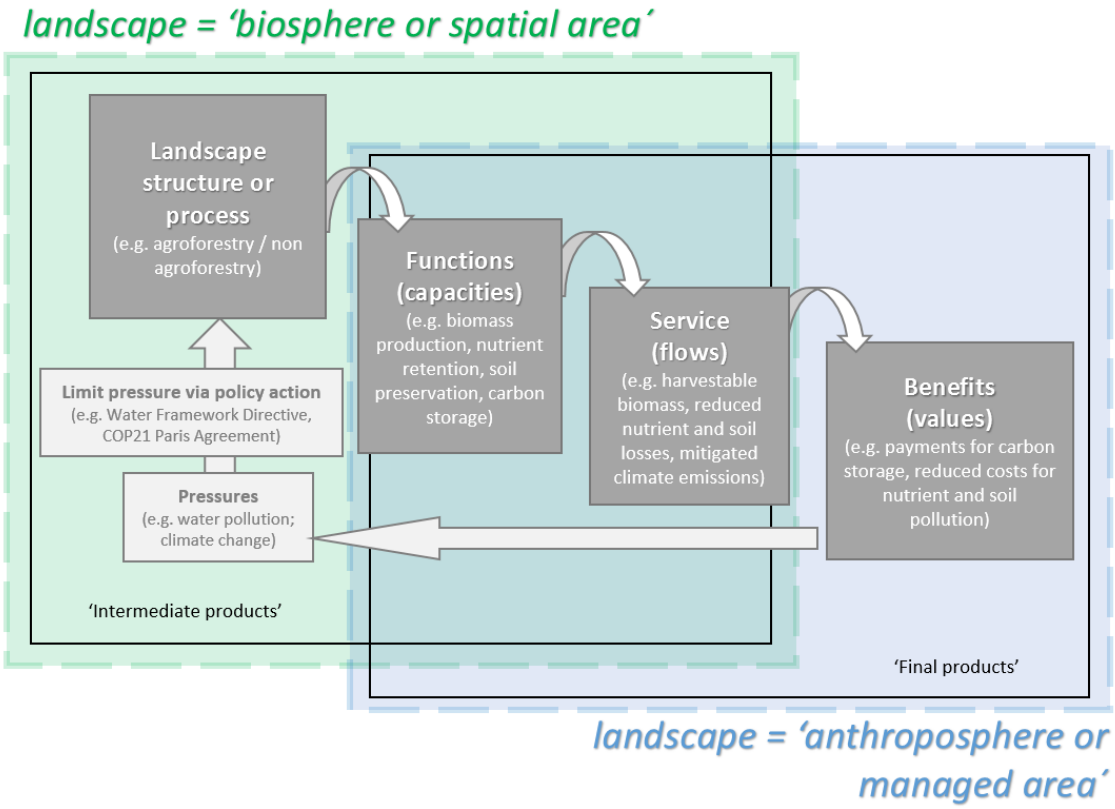

Figure 1: Adapted "Cascade model" from Haines-Young and Potschin (2010) – The relationship between biodiversity, ecosystem function and human well-being for the example of agroforestry systems and the linkage to the different meanings of the landscape term are depicted

As a result of the interlinkage between ES and landscape approaches, the established and validated methods used to investigate landscapes and their processes are also effective for the ES assessment. Regrettably this also includes their limitations.

1.3.3 Ecosystem services in (agricultural) landscapes

Despite the fact that the number of studies linking ES to landscape have increased (Englund et al., 2017), there exists no uniform definition of landscape in meaning, size or configuration (Chapter 1.2). The diverging connotation of "landscape" (as shown in Figure 2) remains valid in the terms "landscape services" and "landscape approaches". "Landscape service" was introduced by Termorshuizen and Opdam (2009) and was thereafter used by Ungaro et al. (2014) and Hainz-Renetzeder et al. (2015) to assess the value of complex landscapes rather than single ecosystems. According to Bastian et al. (2014) "*Landscape services are the contributions of landscapes and landscape elements to human well-being*". In contrast, the term "landscape approach", as presented by Sayer et al. (2013), characterises a stakeholder-driven process to

manage land and achieve social, economic, and environmental objectives, ideally by consensus. This stakeholder involvement was further highlighted by Minang et al. (2014) and FAO (2017) for managing agricultural landscapes.

As one of the main land users in Europe (Eurostat, 2013) and worldwide (FAO, 2018), agriculture and its ES are often addressed in research. Herein, a wide variety of topics, regions and scales are covered. Antle and Stoorvogel (2006), for example, discussed the market and policy dilemma between market driven food, fibre and energy production and societal expectations for non-marketable goods such as clean air and water in the US; in addition, Tscharntke et al. (2005) focussed on the effect of the intensification of agricultural production and the effects on pollinator populations in Europe. Moreover, Van Berkel and Verburg (2014) evaluated the cultural ecosystem services provided by agricultural landscapes.

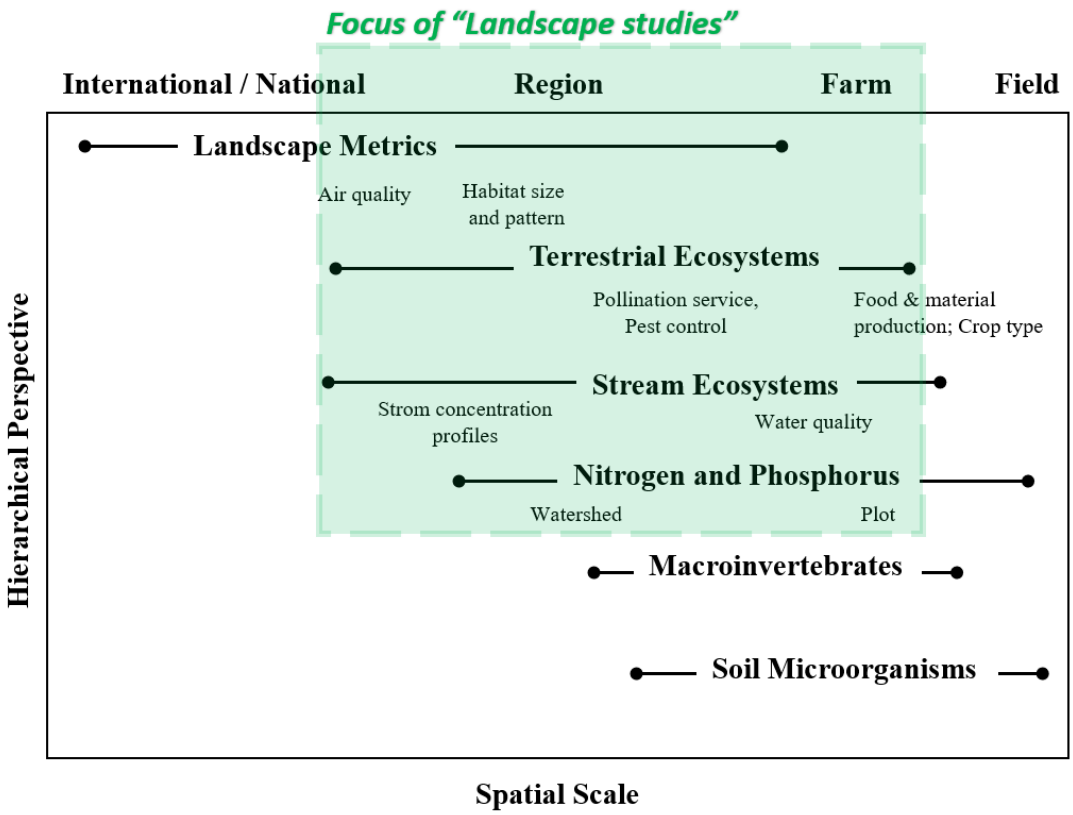

Figure 2: Landscape studies in the context of spatial scales of agricultural ecosystem services (adopted figure by Dale and Polasky, 2007)

Although farming provides multiple services whose priority is food production, it is affected by dis-services (water competition, pest infestation, etc.) and also causes them (habitat losses, nutrient pollutions, etc.) (Zhang et al., 2007). Moreover, studies have identified agriculture as

one of the main drivers of landscape changes (Plieninger et al., 2016; van der Zanden et al., 2016) due to e.g. mechanisation and intensification of production followed by simplification and removal of landscape elements (Biasi et al., 2016). Dale and Polasky (2007) assessed the impact of agricultural practices on ES provision and presented a relation between single ES indicator and pertinent spatial scales. Consequently, they proposed to explore multiple indicators on various scales for valid investigations. Landscapes studies, as investigated by Englund et al. (2017), picked up this point by assessing between 1 and 14 (on average 2.8) ES indicators. Biomass production, climate regulation, lifecycle maintenance and mediation of mass flows constitute the most assessed provisioning and regulating ES (Figure 2).

1.4 Agroforestry

1.4.1 Definition and Classification

Agroforestry systems (AF) were defined by Somarriba (1992) as "*a form of multiple cropping which satisfies three basic conditions: 1) there exist at least two plant species that interact biologically, 2) at least one of the plant species is a woody perennial, and 3) at least one of the plant species is managed for forage, annual or perennial crop production*". A simpler definition was used by the European Commission (2013), according to which agroforestry comprises "*land use systems in which trees are grown in combination with agriculture on the same land*". Moreover, the FAO (2015) defined it 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*".

Traditionally these land use practices served as a primary source of food and resources in subsistence farming and thus were widespread (Nerlich et al., 2013). In Europe, wood pastures in the Mediterranean (Montado, Dehesas), fruit or olive production with undercropping in south and central Europe (Streuobst, Groves) and windbreaks and hedgerows in the coastal areas (Bocage, Knicks) are just some examples of the huge variety of agroforestry systems. They cover around 15.4 million hectares, which equates to approximately 8.8% of European agricultural land. Den Herder et al. (2017) 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). Nonetheless, many of these traditional agroforestry systems are in decline (Eichhorn et al., 2006; Nerlich et al., 2013; Sereke et al., 2016).

Mosquera-Losada et al. (2016) summarised said systems into five categories: (1) silvopastoral systems – woody elements with forage and animal production, (2) silvoarable systems – woody

elements intercropped with annual or perennial crops, (3) hedgerows, windbreaks, riparian buffer strips – lines of woody elements bordering farmland, all three categories on agricultural land, (4) forest farming on forest land and (5) homegardens in urban areas (Figure 3). The sole focus of this study is agroforestry on agricultural land.

(a) Agricultural Land		(b) Forest land	
	**Silvopastoral** Combining trees and shrubs with forage and animal production		**Forest farming** Forested areas used for harvest of speciality crops or pasture
	**Silvoarable** Widely spaced trees and shrubs inter-cropped with annual or perennial crops	**(c) Urban Areas**	
			**Homegardens** Trees / shrubs in urban areas
	**Hedgerows, windbreak, riparian buffer strips** Lines of trees/ shrubs bordering farmland to protect livestock, crops, and /or soil /water quality		

Figure 3: Classification of agroforestry systems according to Mosquera-Losada et al. (2016).

Despite the fact that the variation is huge, agroforestry systems have in common that they offer at least two different kinds of marketable products (food or fodder and timber). Additionally, they are known to provide environmental benefits and ecosystem services (Jose, 2009; Pimentel et al., 1992). Many traditional agroforestry systems have been classified as high nature value and biodiversity systems (McNeely and Schroth, 2006; Oppermann et al., 2012) and are therefore listed in the EU Habitats Directive, receiving protection under the NATURA 2000 network (European Commission, 1992).

According to stakeholders, the key benefits of agroforestry in Europe are the improvement of the environmental value of agricultural land, enhanced biodiversity and habitats, animal health, and landscape aesthetics (García de Jalón et al., 2018a; Rois-Díaz et al., 2018). Labour complexity and intensity, together with administrative burdens, were mentioned as the biggest constraints.

1.4.2 Ecosystem service provided by agroforestry

The role of agroforestry in providing ES at the plot level has been investigated in several studies (e.g. Udawatta et al. 2008; Nair 2012; Alam et al. 2014; Moreno et al. 2016). Torralba et al. (2016) summarised the key ES as: (1) timber, food and biomass production, (2) soil fertility and nutrient cycling, (3) erosion control and (4) biodiversity provision. Recently their ability to store carbon has been highlighted (Hart et al., 2017).

1.4.2.1 Timber, food and biomass production

While the biggest advantage of AF is its broad product portfolio, consisting of timber, food and fodder production, due to this interspecific complexity and the long-term effects, a yield assessment is challenging, and modelling approaches are often used to quantify the outcomes. Examples in this regard are YieldSAFE (van der Werf et al., 2007), a process-based parameter-sparse dynamic model, Hi-sAFe (Talbot, 2011), a 3-D process-based model, WoodPaM (Gillet, 2008), ALWAYS (Bergez et al., 1999) for silvopastoral systems, and ESAT-A (Tsonkova et al., 2014) for alley cropping systems in particular.

Depending on the system, the rotation length of trees, and the geographical location, huge variations were reported. While Van Vooren et al. (2016) predicted a reduced biomass production in Dutch agroforestry systems, Graves et al. (2007), Palma et al. (2007) and Sereke et al. (2015) found higher production levels.

1.4.2.2 Nutrient emissions

Nair et al. (2007) and Jose (2009) showed that agroforestry can help reduce nutrient losses by 40 and 70%, respectively. Moreover, López-Díaz et al. (2011) demonstrated, through greenhouse experiments, that trees had a higher root density and a deeper root horizon, which led to a higher uptake of nitrate and a reduction of nitrate leaching of 38 to 85%. Recently, Hartmann and Lamersdorf (2015) contrasted agroforestry systems with short rotation coppies in Germany and found a medium leaching rate for agroforestry (4.4 kg ha^{-1} yr^{-1}) compared to poplar coppies (2.5 kg ha^{-1} yr^{-1}) and willow coppies (22.3 kg ha^{-1} yr^{-1}). Additionally, Udawatta

et al. (2002) and Anderson et al. (2009) demonstrated that agroforestry and buffer trees reduced runoff water flows and consequently nutrient surface losses.

1.4.2.3 Erosion control

Another aspect of decreased runoff waters is the effect of soil preservation. McIvor et al. (2014) summarised the effect of agroforestry systems on arable land and pastures for soil preservation. For example, García-Ruiz (2010) and Durán Zuazo and Rodríguez Pleguezuelo (2008) investigated soil erosion in Spain and showed that a higher vegetation cover was connected to reduced runoff waters and erosion. In addition, Reubens et al. (2007) focussed on the positive effects of vegetation roots, especially of woody species, on slope stabilisation and soil erosion. Beside their hydrological and mechanical soil binding capacity, the morphology of the roots and their architecture were defined as key factors for soil fixing efficiency.

1.4.2.4 Biodiversity

Due to their heterogeneous composition, with diverse vertical and horizontal structures, agroforestry systems provide multiple habitats for various flora and fauna species. For example, studies by Moreno et al. (2016) in Spain demonstrated that large extended agroforestry landscapes (e.g. Dehesas) with a lot of coexisting habitats promote the incidence of different species. Similar observations exist for fruit orchard landscapes in temperate Europe, which have also been shown to harbour high species richness and, in particular, specialised species such as orchard birds (e.g. Birrer et al. 2007; Bailey et al. 2010). In Atlantic hedgerow landscapes (e.g. Bocage) Lecq et al. (2017) identified the ground refuges as important habitats and especially microhabitats for vertebrate and invertebrate species groups.

1.4.2.5 Climate regulation and carbon storage

Zomer et al. (2016) showed the important effect of trees on agricultural land for global carbon storage. They estimated the carbon storage of agroforestry systems in Europe to be around 4 t C ha^{-1}*. It is seen as the land-use option with the greatest potential for climate mitigation and adaptation in the agricultural sector in Europe (Alig et al., 2015; Hart et al., 2017) and worldwide (Smith et al., 2008). Cardinael et al. (2015), Kim et al. (2016) and Nabuurs and Schelhaas (2002) demonstrated that agroforestry had the ability to capture carbon in tree and root biomass and additionally increase soil carbon stock.

* 1 t = 1 Mg = 10^9 gm; 1 gm C = 3.67 gm CO_2eq

1.5 Research questions and hypotheses

Against the above-presented background of global trends and political framework (Chapter 1.1) a question arises regarding the impact which agroforestry systems have at the landscape scale (Chapter 1.2) on ecosystem services provision (Chapter 1.3). Figure 4 visualises the integration of agroforestry systems as part of the landscape using the ecosystem service framework as assessment tool in regards to the social and political context. Therefore, the aim of this thesis is to evaluate the effect of temperate agroforestry systems on ecosystem service provision at the landscape level.

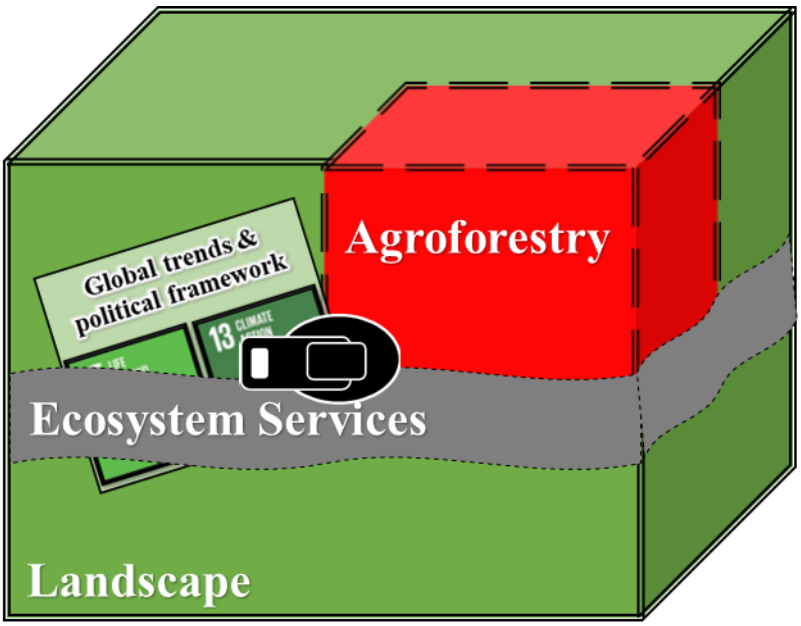

Figure 4: Conceptual background of the interlinkage between agroforestry systems and their impact on landscapes analysed using the ecosystem service framework.

This leads to the three main research questions, which will be answered by testing the following hypotheses (HP):

1. Does the provision of ecosystem services differ in landscapes with agroforestry compared to landscapes dominated by agriculture?

HP 1: Agroforestry systems provide multiple ES and have an overall positive effect on conventional agricultural farming at a plot level (Alam et al., 2014; Torralba et al., 2016). Hypothesising that this positive effect of agroforestry radiates at the landscape level results in an overall higher provision of provisioning and regulating ES from landscapes with agroforestry systems compared to landscapes with conventional agriculture.

HP 2: The beneficial impact of agroforestry at a landscape level can be verified for various temperate agroforestry systems in Europe (Moreno et al., 2018; Pantera et al., 2018).

2. Is this ecosystem service provision related to economic and environmental benefits within these landscapes?

HP 3: Valuing provisioning and regulating ES increase the profitability of landscapes with agroforestry and agro-ecological land management systems compared to agricultural landscapes (Alam et al., 2014; Zander et al., 2016).

3. Can agroforestry systems significantly contribute to European climate targets of zero-emission agriculture?

HP 4: Agroforestry systems have a high climate change mitigation potential (in combination with other environmental and production benefits) in Europe (Hart et al., 2017).

These hypotheses were tested within the research project AGFORWARD – AGroFORestry that Will Advance Rural Development* founded by the European Union's Seventh Framework Programme for Research and Technological Development (FP7) between January 2014 and December 2017. The project aimed to promote agroforestry practices in Europe to achieve advanced rural development and social and environmental enhancement. One of the main objectives was to increase the understanding of agroforestry systems and to identify, develop and demonstrate ecosystem services benefits in Europe. This was achieved using 12 traditional agroforestry landscapes all over Europe as examples. They were selected on the basis of: (1) their biogeographical region (Continental, Mediterranean, Atlantic and Boreal) and (2) the type of agroforestry systems present. To evaluate differences between agriculture and agroforestry landscapes, in each region eight landscape test sites (each 1 km^2) were selected. Four LTS were dominated by agroforestry systems (AF-LTS), while the other four were dominated by agriculture (NAF-LTS).

The first hypothesis was investigated in a single case study region (the Swiss cherry orchards); to this end, a landscape ES evaluation toolkit was elaborated on, which brought together and interlinked state-of-the-art models for the evaluation of major provisioning and regulating ES.

* European Union's Seventh Framework Program for research, technological development and demonstration under grant agreement no 613520

To test the second hypothesis, the ES evaluation toolkit was applied to six case study regions form Atlantic, Mediterranean, and Continental Europe. The third hypothesis was based on the results of 12 case study regions. Finally, the fourth hypothesis assessed the potential impact of agroforestry on European agricultural land at the continental level.

Table 2 lists the regions, the evaluated systems and the addressed hypothesis. Figure 5 shows the location of the case study regions and their associated research questions and hypotheses.

Table 2: Research question (RQ) and hypotheses (HP) linked to study regions and their dominating agroforestry (AF-LTS) and agricultural systems (NAF-LTS)

RQ	HP	Country	Abb.	LTS type	System	Biogeographical region
RQ 1 RQ2	**HP 1 HP2 HP3**	Switzer-land	CH1	AF-LTS	Fruit orchard (Cherry, *Prunus avium* L.)	Continental
				NAF-LTS	Open pasture and arable farming	
RQ 1 RQ2	**HP2 HP3**	Portugal	PT	AF-LTS	Montado - Wood pasture (Cork oak, *Quercus suber* L.)	Mediterranean
				NAF-LTS	Open pasture	
RQ 1 RQ2	**HP2 HP3**	Spain	ES1	AF-LTS	Dehesa - Wood pasture (Holm oak, *Quercus ilex* L.)	Mediterranean
				NAF-LTS	Open pasture	
RQ 1 RQ2	**HP2 HP3**	Switzer-land	CH2	AF-LTS	Wood pasture (Spruce, *Picea abies* L.)	Continental
				NAF-LTS	Open pasture	
RQ 1 RQ2	**HP2 HP3**	Spain	ES3	AF-LTS	Chestnut soutos (*Castanaea sativa* Miller)	Atlantic
				NAF-LTS	Open pasture and arable farming	
RQ 1 RQ2	**HP2 HP3**	United Kingdom	UK	AF-LTS	Hedgerow landscape with arable farming (mixed species)	Atlantic
				NAF-LTS	Arable farming	
RQ2	**HP3**	Greece	GR	AF-LTS	Intercrop olive groves (*Olea europaea* L.)	Mediterranean
				NAF-LTS	Intensive olive groves (*Olea europaea* L.)	
RQ2	**HP3**	Spain	ES2	AF-LTS	Intercrop oak (Holm oak, *Quercus ilex* L.)	Mediterranean
				NAF-LTS	Arable farming	
RQ2	**HP3**	Romania	RO	AF-LTS	Wood pasture (Common Oak, *Quercus robur* L.)	Continental
				NAF-LTS	Open pasture	
RQ2	**HP3**	Germany	GE	AF-LTS	Hedgerow landscape with arable farming (mixed species)	Continental
				NAF-LTS	Arable farming	
RQ2	**HP3**	France	FR	AF-LTS	Bocage - Mixed arable-pasture systems fenced by hedgerows (mixed species)	Atlantic
				NAF-LTS	Mixed arable-pasture systems	
RQ2	**HP3**	Sweden	SW	AF-LTS	Wood pasture (Common Oak, *Quercus robur* L.)	Boreal
				NAF-LTS	Open pasture	
RQ3	**HP4**	European Union 27 + Switzerland	EU+		Sixty-four potential novel agroforestry systems	

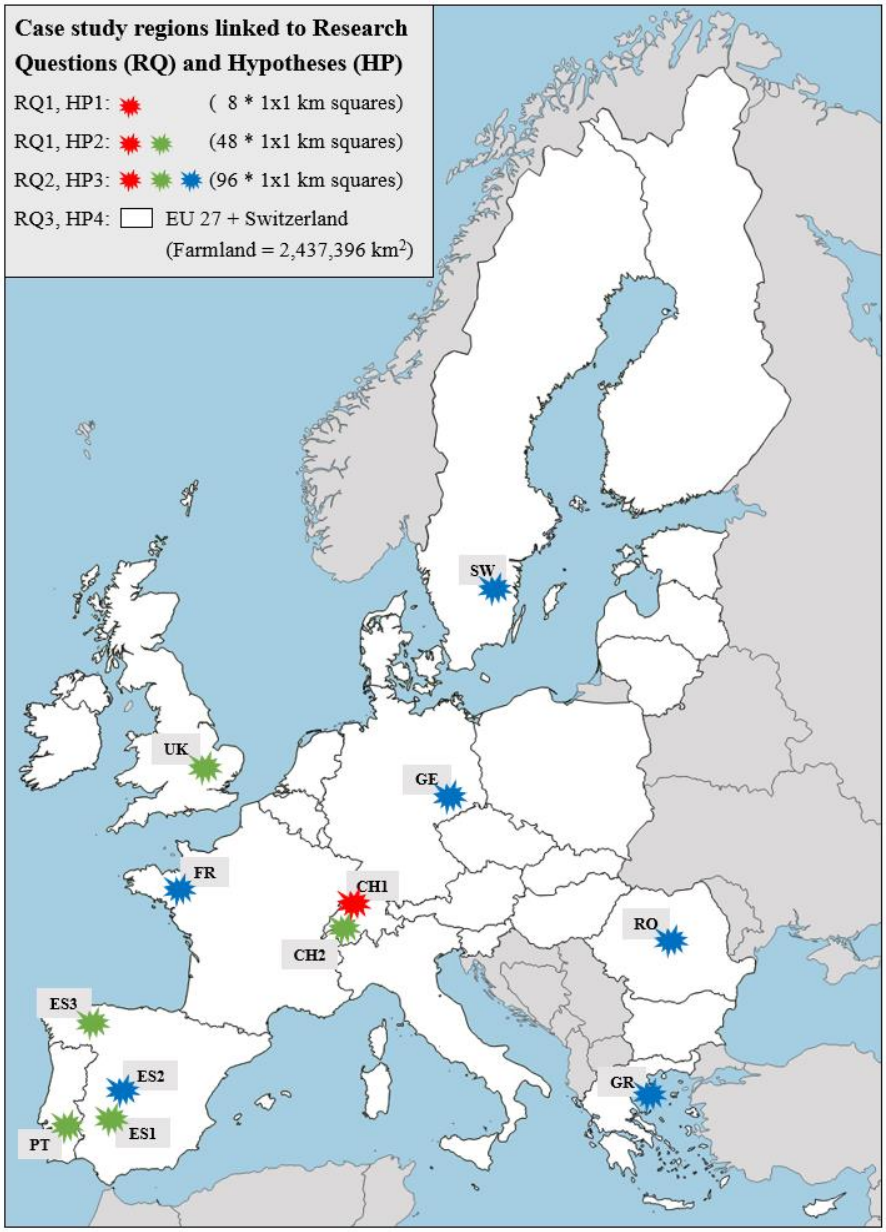

Figure 5: Spatial location of case study regions and associated research questions (RQ) and hypotheses (HP)

The thesis is structured in line with the above-presented hypotheses. Chapter 2 develops a methodological approach to assess and comprehensively quantify a bundle of ES related to agroforestry systems. In Chapter 3 this approach is transferred to six traditional agroforestry landscapes in Europe. The economic evaluation of the ecosystem services is presented in Chapter 4, while Chapter 5 evaluates the potential contribution of agroforestry to European agricultural climate targets. Finally, Chapter 6 synthesises the thesis into the main findings and an outlook.

1.6 Contribution of the first author and the co-authors

The presented work was part of the AGFORWARD research project, which involved 25 research, farming, and governance organisations across Europe. The collaboration and interlinkage between the partners, the regions, and their research was one of the key targets within the project. Against this background, the following chapters present the result of collaborative work by several authors from different countries and organisations, as indicated at the beginning of each chapter. Specifically, the field mapping and the validation of the outcomes were conducted by local partners for their specific case study region.

A transparent presentation of the individual contribution of the first author and the co-authors follows the recommendations of the Swiss Academies of Arts and Science "Authorship in scientific publications" (SwissAcademies, 2013). Accordingly, an author (a) made a substantial contribution to the planning, execution, evaluation, and supervision of the research, (b) was involved in writing the manuscript, and (c) had approved the final version of the manuscript. Scoring systems are not widely used; hence an example was given by Kosslyn (2002). His 1.000-Point System values the idea, the design, the implementation, the conduction of the experiment, the data analysis, and the writing. Table 3 applies both systems to the present work.

Table 3: Contribution of first author and co-authors to the individual chapters

Chapter	Swiss Academies	1.000-Point-System	First Author	Co-Authors	Scoring
1	All tasks		x		
	Total		100%		
2	Substantial contribution to research	Idea		x	250
		Design	x	x	100
		Data collection	x	x	100
		Spatial Modelling	x		100
		Data analysis	x		200
	Writing manuscript	Writing	x	x	150
	Approving final		x	x	100
	Total		55%	45%	1,000
3	Substantial contribution to research	Idea		x	250
		Design	x		100
		Data collection	x	x	100
		Spatial Modelling	x		100
		Data analysis	x		200
	Writing manuscript	Writing	x	x	150
	Approving final		x	x	100
	Total		64%	36%	1,000

4	Substantial contribution to research	Idea	x		250
		Design	x		100
		Data collection	x	x	100
		Spatial Modelling	x		100
		Data analysis	x		200
	Writing manuscript	Writing	x	x	150
	Approving final		x	x	100
	Total		85%	15%	1,000
5	Substantial contribution to research	Idea	x	x	250
		Design	x	x	100
		Data collection	x	x	100
		Spatial Modelling	x		100
		Data analysis	x		200
	Writing manuscript	Writing	x	x	150
	Approving final		x	x	100
	Total		81%	19%	1,000
6	All tasks		x		
	Total		100%		

Chapter 2

Landscape-scale modelling of agroforestry ecosystems services in Swiss orchards: A methodological approach

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Abstract:

Context: Agroforestry systems in temperate Europe are known to provide provisioning and regulating ecosystem services (ES). Yet, it is poorly understood how these systems affect ES provision at a landscape scale in contrast to agricultural practises.

Objectives: This study aimed at developing a spatially explicit model to assess and quantify bundles of ES provided by landscapes with and without agroforestry 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 1km x 1km, 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 landscapes with agroforestry. The differences were found to be statistically significant for the indicators annual biomass yield, groundwater recharge rate, nutrient retention, 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 scenario at landscape scale.

Keywords:

biodiversity; biomass production; carbon sequestration; erosion; groundwater recharge; nitrate leaching; pollination

2.1 Introduction

Agroforestry systems are traditional man-made agricultural land use practices, combining woody perennials with agricultural crops and / or animals to provide food, fodder, and timber from the same field at the same time (European Commission, 2013a). In addition to this range of products, the systems offer many environmental benefits and help conserve autochthonous biodiversity (Moreno et al., 2018). However, the specialisation and mechanisation of agricultural production over the last decades has discouraged farmers to maintain agroforestry systems (Nerlich et al., 2013).

In order to appreciate all "benefits people obtain from ecosystems", the Millennium Ecosystems Assessment (MEA) developed the ecosystem service (ES) framework in 2003, which valued provisioning, regulating and cultural services (MEA 2003). Since then, research has aimed to map and assess these ES (e.g. Syswerda & Robertson 2014; Maes et al. 2016), more recently also accounting for their spatial allocation and the effect of spatial patterns on bundles of ES (Crouzat et al., 2015). Spatial pattern, especially in agricultural landscapes, is a result of land cover, land management and topographic conditions (Verburg et al., 2013) and is directly related to the function and supply of ES (Englund et al. 2017). Notwithstanding, different ES, goals and stakeholder interests relate to different ecosystems and spatial scales (Hein et al., 2006). Managing these multiple goals and various demands on land requires an understanding of landscapes (Jones et al., 2013). Against this background, Termorshuizen and Opdam (2009) developed the "landscape service" concept, which assesses the value of complex landscapes rather than of single ecosystems. Scherr et al. (2012) underlined that only an integrated landscape management will sustainably fulfil the multiple future purposes demanded by stakeholders.

In this context, the multifunctionality of agroforestry systems could play a key role in landscape and agricultural management. They provide marketable products and deliver comparatively more provisioning and regulating ES in comparison to agricultural and forest plots (Alam et al., 2014). Torralba et al. (2016) mention (1) timber, food, and biomass production, (2) soil fertility and nutrient cycling, (3) erosion control and (4) biodiversity provision as ES of major importance. However, the existing investigations of ES provision by agroforestry are mainly restricted to single services and / or to field scale (Pumariño et al., 2015; Udawatta et al., 2008).

European agroforestry can be sub-divided into temperate and Mediterranean agroforestry systems (Eichhorn et al., 2006). Currently European agroforestry covers around 15.4 million

hectares, 79% of which are in the Mediterranean parts of Europe; in Spain, Portugal, southern France, Italy, Greece, and Romania (den Herder et al., 2017). While in former times temperate Europe had a remarkable amount of agroforestry land, the majority of fruit orchards and wind breaks were transformed into pure agricultural areas (Nerlich et al., 2013; Sereke et al., 2015). More recently, however, the awareness of the benefits of agroforestry systems as ES providers is increasing and both, farmers and policy makers are seeking for ways to re-introduce trees in agricultural landscapes also in temperate Europe (Garibaldi et al., 2017; Maes et al., 2015).

There is a need, therefore, for a spatially explicit and systematic assessment of how temperate agroforestry systems affect the ES provision of landscapes and influence landscape services in comparison to agricultural land use.

Until now, the evaluation of bundles of ES mostly rested on expert grading approaches (e.g. Burkhard et al., 2009; Jacobs et al., 2015). To be less dependent on expert opinion, our first objective was to develop a methodology to assess and comprehensively quantify a bundle of ES with a semi-quantitative approach at the landscape scale through a combination of field investigations and modelling. Whilst the model involves existing and well established individual algorithms for the evaluation of the above-mentioned ES at the plot scale, this is the first time that they are applied at the landscape scale and in combination.

Our study focussed on Swiss cherry orchards because traditional fruit orchards are one of the major agroforestry system of temperate Europe (e.g. Herzog 1998). Our second objective was to test the hypothesis that the ES provision of agricultural landscapes will differ from landscapes with agroforestry plots. In undertaking this evaluation, we first selected indicators that could be used (1) to address the differences in performance of agroforestry and agricultural systems, (2) were relevant for farmers, policy makers, and society, and (3) could be used as steering wheels for landscape management. Then, algorithms for quantifying these indicators were identified, tested, adapted, and applied to compare ES provision between agroforestry (AF) and non-agroforestry (NAF) landscapes.

2.2 Data and Methods

2.2.1 Study area

The study was conducted in traditional high-stem cherry orchards in north-western Switzerland. The region is known for a long tradition in cherry production, due to the comparatively mild climate where late frost is infrequent. The case study region comprises seven municipalities and is typical for many hilly regions of temperate Europe (Figure 6). Forestry and farmland are

the main land-uses, agroforestry is present on 5% of the area and 8% are covered by settlements. Most farming enterprises are mixed farms with combinations of arable crops and animal husbandry (mostly cattle for milk and meat production) and some fruit production. With an average farm size of 24ha, the farms are slightly larger than the average Swiss farm (around 20ha, BLW 2017).

The evaluated agroforestry system consists of around 80 cherry trees ha^{-1} on grassland. The trees are heterogeneous in age and provide cherries and timber. The cherries are harvested for liquor, tinned food, or direct consumption. The grassland is used as hay, silage or pasture. Traditionally, cherry orchards were present on most farms but more recently, cherry production with standard fruit trees is in decline due to high labour costs and the invasive fruit fly *Drosophila suzukii*.

Cherry Orchard, Switzerland	
Municipalities	Büren, Gempen, Hochwald, Lupsingen, Nuglar-St. Pantaleon, Seewen, Seltisberg
Area	49.89 km^2
Temperature Avg °C	7.7 °C
Precipitation	800 - 1000 mm
Elevation	430 - 670 m
Soil	fine
Land use	43 % Non-Agroforestry, 44 % Forestry, 5% Agroforestry, 8% others
Agriculture	1,972 ha farmland 83 farmers 1,522 LU (mostly cattle)
Livestock intensity	0.77 LU ha^{-1} farmland
Agroforestry system	80 cherry trees ha^{-1} + grassland
Products	Cherries for liquor, tinned food or direct consumption Grass as fodder for cattle (hay, silage or pasture) Timber

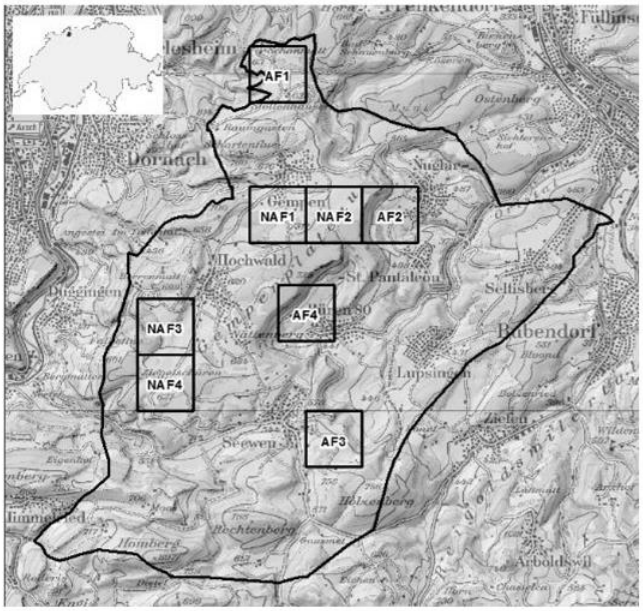

Figure 6: Profile of the cherry orchard case study region, Switzerland (LU: livestock unit). AF 1 - 4: Landscape test sites of 1km x 1 km with a high share of cherry orchards; NAF 1 – 4: Landscape test sites dominated by agricultural land use.

2.2.2 Selection of Landscape Test Sites

We subdivided the case study area into the broad land cover categories forestry, agroforestry and agriculture (mainly arable). In both, the agroforestry (AF) and agricultural (non-agroforestry, NAF) sector, we randomly picked four landscape test sites (LTS) of 1 x 1km, resulting in eight LTS altogether. In each LTS habitats and trees were mapped in the field during spring 2015 and 2016. For grassland, percentage cover of grass, clover, and herbs was recorded. For woody perennials, the location, tree species, height, and structure were recorded during field surveys. Single trees and AF trees were digitized from aerial photographs and classified by crown diameter as small (young), medium (middle age), and large (old). The location of arable and other land was identified and mapped. All information was combined in a habitat map and digitized using ArcGIS 10.4. It should be noted that it was not possible to find entire LTS under AF and NAF and therefore each LTS included a mix of arable, grass, forestry, agroforestry, and other (urban) land covers. However, agroforestry dominated the land cover in the AF LTS (31-55%) and arable land dominated the land cover in the NAF LTS (58-72%).

2.2.3 ES assessment

A range of indicators were selected to compare ES delivery in the AF and NAF LTS, based on provisioning and regulating services listed in the Common International Classification of Ecosystem Services (CICES) version 4.3 (Haines-Young and Potschin, 2013). The selected ES indicators were Annual Biomass Yield and Biomass Stock (for the ES biomass production), Groundwater Recharge Rate (for the ES groundwater recharge), Nitrate Leaching (for the ES nutrient retention), Soil Erosion (for the ES soil preservation), Annual Carbon Sequestration and Carbon Stock (for the ES carbon storage), Pollination Services, Flowering Resources, Ground and Cavity Nesting Resources for solitary bees, the Simpson Diversity Index, the Share of Semi-Natural Habitat, and the Richness of Semi-Natural Habitat Types (for the ES habitat and gene pool protection). A spatially explicit ES evaluation model was developed, which comprised the fifteen selected indicators and accounted for their interaction (Figure 7). In order to consider the spatial dependence, location-dependent variables such as habitat map, soil map, digital elevation model, and climate conditions were used to calculate each indicator. Model outcomes were ES maps (resolution 2 x 2m) (Figure 8a), wherein each pixel contained the information for all indicators and specified the relationship to that specific location. The indicator values were then aggregated at the LTS scale and quantified as mean per hectare values for the whole LTS area. In the following sections the approaches are summarized, a detailed description of the models can be found in the Annex I.

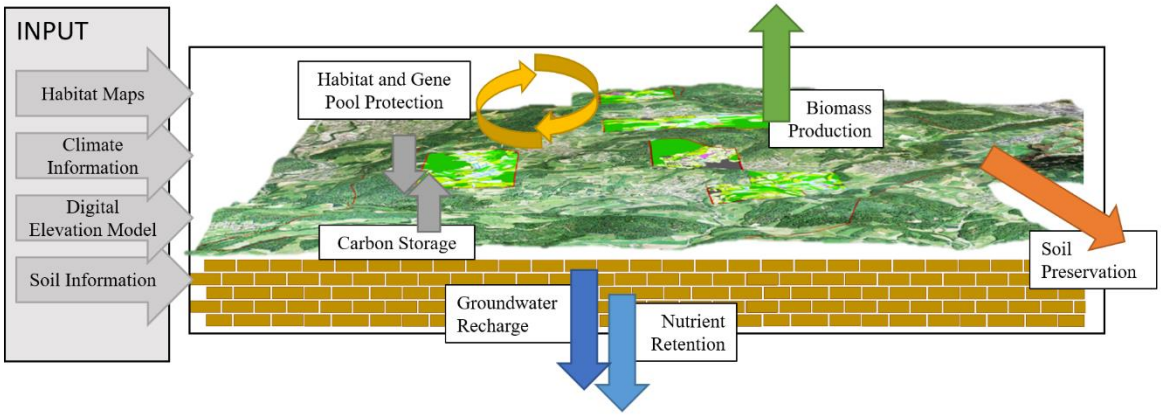

Figure 7: Conceptual background of the model

2.2.3.1 Biomass production

Biomass production was modelled using the EcoYield-SAFE model (Palma et al., submitted) for agroforestry systems, and the Swiss statistical data for agricultural and forest production (AGRIDEA and BLW, 2017; BAFU, 2013; BAFU and BfS, 2015a; Brändli, 2010). The biomass stock value at any one time [unit: t DM ha^{-1}], and the annual biomass yield [unit: t DM ha^{-1} yr^{-1}] were assessed separately for agricultural, forestry, and agroforestry systems. The annual values represent the status quo as mapped in the field. Herein, young trees provided annual prunings and cherries, while old trees provided timber and cherries. The accumulated biomass stock represented the sum of all perennial biomass. To enable comparison between the LTS, no distinction was made regarding the type and quality of biomass. This assumption is compliant with previous agroforestry research by e.g. Tsonkova et al. (2014) and Fader et al. (2015).

2.2.3.2 Groundwater recharge

Water flows to groundwater are directly linked to land cover, land management and landscape structure. Based on the general water equation, the water flows were modelled by using FAO's CROPWAT 2.0 for crop performance indices (Allen et al., 1998) in combination with the spatial components of MODIFFUS 3.0 method (Hürdler et al. 2015). Our focus was on the amount of groundwater recharge in percent of the total precipitation [unit: % of precipitation].

2.2.3.3 Nutrient retention

The focus of this ES was on nitrogen leaching and phosphorus losses. The nutrient loss assessment was based on MODIFFUS 3.0, an empirical model for nitrate and phosphorus losses in Switzerland (Hürdler et al. 2015) and was expressed in kg N ha^{-1} yr^{-1} and kg P ha^{-1} yr^{-1}.

2.2.3.4 Soil preservation

A major indicator of effective disturbance regulation is soil erosion. This indicator was assessed using the Revised Universal Soil Loss Equation (RUSLE) (Renard et al., 1997) for the annual soil loss in tonnes per hectare [unit: t soil ha^{-1} yr^{-1}].

2.2.3.5 Carbon storage

Our assessment of biomass carbon storage was based on the produced above and below ground biomass estimated in EcoYield-SAFE. In addition, we used Yasso07 to model soil organic carbon (Liski et al., 2005). The outcomes were divided into the annual carbon sequestration [unit: t C ha^{-1} yr^{-1}] and the carbon stock [unit: t C ha^{-1}].

2.2.3.6 Habitat and gene pool protection

The pollination indicator was assessed using the Lonsdorf model (Lonsdorf et al., 2009). It estimates the habitat nesting suitability, the habitat flowering suitability, and the reachability between these two. The nesting capacity was evaluated for both ground and cavity nesting wild bee species. Ground nesting facilities were mapped in the field. Cavity nesting potential was assumed to be present in all habitats with woody elements. The flowering potential was mapped using the quantity of clover and herbs in grasslands, crops pollinated by insects, and blossoming trees. To model the pollinator index for a range of pollinators, three moving corridors (100, 350, 500m) were computed for the two nesting types.

The structural diversity of agroforestry systems was evaluated by the Simpson Diversity Index (SIDI, no unit), the share of semi-natural habitat (SoSNH, in percent), and the richness of the semi-natural habitat types (ToSNH, number). The indicators were computed from the habitat maps. They indicated relative levels of habitat and – potentially – species diversity in the case study region.

2.2.4 Spatial and statistical analysis

To compare ES provision from AF and NAF landscapes, all ES were modelled for all land use types and then aggregated in relation to their spatial extent (indicators for biomass, carbon) or directly computed at the LTS scale (indicators involving lateral processes, i.e. soil erosion and habitat indicators relating to landscape composition). The spatial analysis was developed in SAGA System for Automated Geoscientific Analyses (Conrad et al. 2015) and ESRI ArcGIS10.4 (Environmental Systems Resource Institute 2016). The statistical analyses were performed as an ANOVA in R (R Development Core Team, 2016) determine whether significant differences in ES delivery existed between the AF and NAF LTS.

2.3 Results

2.3.1 LTS inventory

Altogether 23 different habitat types and 8,189 trees were recorded across the eight LTS. Figure 8a shows the results of the habitat mapping presenting all eight LTS.

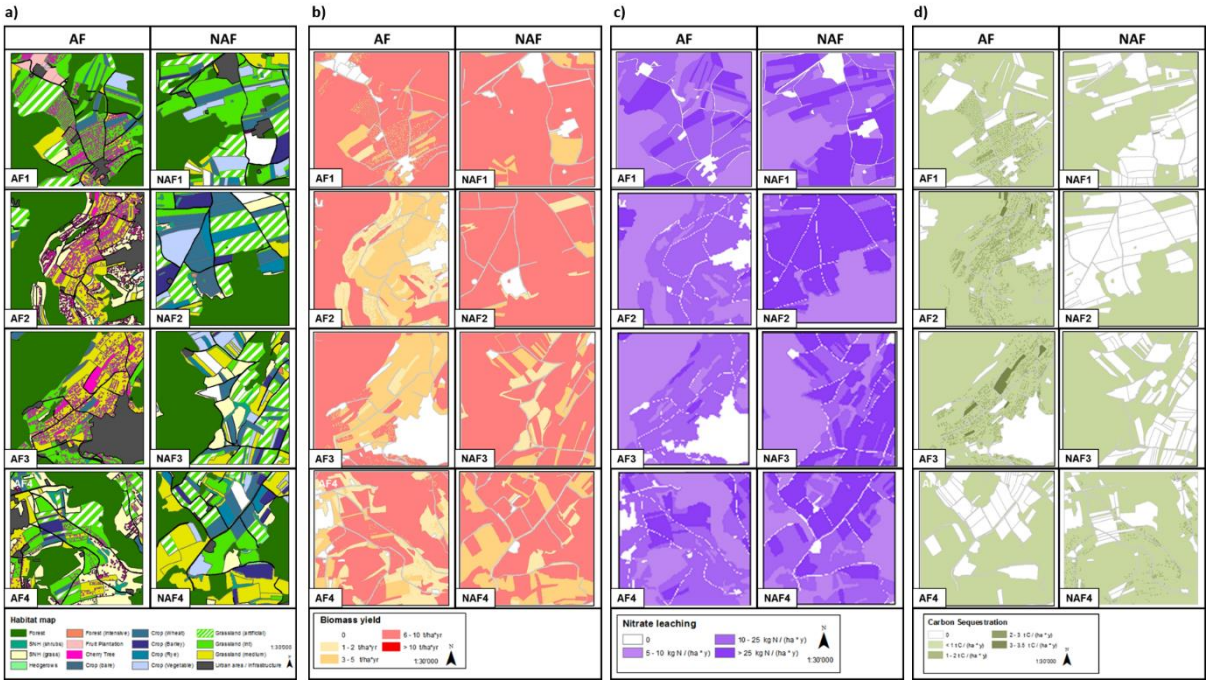

Figure 8: Habitat maps (a), annual biomass yield [t ha-1 yr-1] (b), nitrate leaching [kg N ha-1 yr-1] (c) and annual carbon sequestration [t C ha-1 yr-1] (d) of landscape test sites [LTS] grouped by land cover categories into agroforestry (AF) and non-agroforestry (NAF) sites

2.3.2 Biomass production

The modelled annual biomass yields are shown in Figure 3b for the eight LTS. Across the LTS, mean annual biomass yields were found to be greater in NAF (6.5 t ha^{-1}) landscapes than in AF landscapes (4.6 t ha^{-1}). This effect was statistically significant ($p < 0.01$). However, in contrast, the biomass stock tended to be greater in AF LTS due to the tree biomass.

2.3.3 Groundwater recharge

In the AF LTS, on average 53,6% of the precipitation were allocated to evapotranspiration and 1.8 % were removed from the area as surface runoff whilst 44.7 % percolated into the soil. In NAF LTS the overall fate of precipitation was comparable, with evapotranspiration accounting for 48.7%, surface runoff for 2.3%, and groundwater recharge for 49.1%. The average groundwater recharge rate was significantly lower in AF LTS (44.6%) than in NAF LTS (49%) ($p<0.025$).

2.3.4 Nutrient retention

The assessment of nitrate leaching (Figure 3c) showed relatively high losses of nitrates associated with LTS with larger arable areas, such as NAF2 and NAF3 (>25 kg N ha^{-1} yr^{-1}). The overall average nitrate leaching was 13.8 kg N ha^{-1} yr^{-1} in NAF LTS, and significantly higher ($p<0.008$) than in AF LTS (7.6 kg N ha^{-1} yr^{-1}). The phosphorus loss in both AF and NAF LTS was below 1 kg P ha^{-1} yr^{-1} and is no longer accounted for.

2.3.5 Soil preservation

The average soil erosion was 1.88 t ha^{-1} yr^{-1} in AF and 1.46 t ha^{-1} yr^{-1} in NAF LTS. These differences were not found to be statistically significant between the two types of landscapes.

2.3.6 Carbon storage

The mean annual carbon sequestration rate was 0.49 t C ha^{-1} yr^{-1} in NAF and 0.75 t C ha^{-1} yr^{-1} in AF LTS, which was significantly higher ($p < 0.01$). The maps in Figure 3d show that this effect was largely due to the high shares of arable land in NAF landscapes, such as found in NAF1, 2, and 3. On the other hand, AF landscapes, such as AF1, 2, and 3, showed relatively high annual carbon sequestration rates associated with the agroforestry habitats. The mean carbon stock was also relatively high in AF LTS at 59.6 t C ha^{-1} compared to 51 t C ha^{-1} in NAF LTS but the differences were not found to be statistically significant.

2.3.7 Habitat and gene pool protection

The AF LTS provided greater resources for pollinators. A mean area of 66.3ha in the AF LTS was mapped as potential habitats for ground nesting solitary bees and bumble bees, 44.8ha for cavity nesting solitary bees and bumble bees, and 21.8ha provided flowering potential. In NAF LTS these figures were lower, with 46.2ha having ground nesting potential, 31.6ha having cavity nesting potential, and 14.3ha providing flowering potential. Yet, the differences were statistically significant only for flowering resources ($p<0.05$).

Within a radius of 100m around a nesting facility, results showed that a larger area of land could be reached by pollinators in AF LTS (97.5 % for cavity nesting species, 98.8 % for ground nesting species) than in NAF LTS (84 % and 93 %, respectively). For cavity nesting species, these differences were significant ($p<0.1$), but not for ground nesting species. At flying distances of 350 m and more, the total area could be accessed by both cavity and ground nesting species.

The assessment of habitat richness was based on the landscape metrics SIDI, SoSNH, and ToSNH. The habitat diversity indicator SIDI ranged from 0.82 to 0.88 in AF, 0.85 to 0.89 in

NAF, and was similar across all the LTS. For the other two indicators, the AF LTS showed higher values. The share of semi-natural habitats, SoSNH, was much greater in AF LTS than in NAF LTS. This difference was highly significant ($p > 0.001$). The number of semi-natural habitat types ToSNH was between 35 to 84 in AF LTS and between 16 to 35 in NAF LTS.

2.3.8 Summary of indicator values

Figure 9 provides a summary of the results using normalized indicator values between -1 (for losses) and 1 (for gains). Statistically significant differences between the AF and NAF LTS, and p values are shown for each of the indicators.

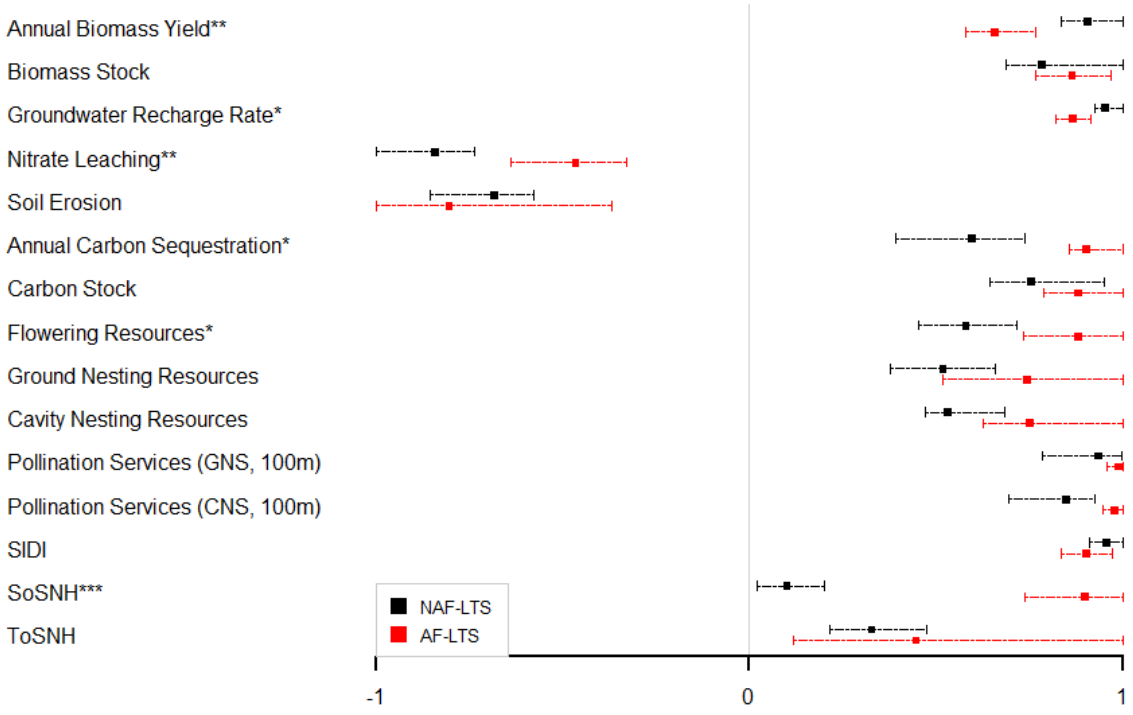

*Figure 9: Summary of the normalized indicators [-1,1] grouped into agroforestry (AF) and non-agroforestry (NAF) landscape test sites normalized to 1 for gains, and -1 for losses (Nitrate Leaching and Soil Erosion) [GNS: Ground Nesting Species, CNS: Cavity Nesting Species, SIDI: Simpson's diversity index, SoSNH: Share of semi-natural Habitat, ToSNH: Richness of semi-natural Habitat; ***: p<0.001 **: p<0.01, *: p< 0.05]*

2.4 Discussion

The study was carried out to develop a spatially explicit model that can be used to evaluate bundles of ES. ES assessment has previously taken place at broad national scales or it was limited to the field scale (Mouchet et al., 2017; Tsonkova et al., 2014). At our intermediate (landscape) scale, a considerable level of detail is needed to account for spatial effects of tree and crop interaction in agroforestry, while on the other hand the methodology has to be balanced

between model complexity, data requirements, and total error (see e.g. Schröter et al. 2014). At this scale, agroforestry assessment itself and their impact on landscape, could be evaluated.

Our second objective was to test the hypothesis that agroforestry and agricultural landscapes provide different quantities of provisioning and regulating ES. We found significant differences for the ES indicators annual biomass yield, groundwater recharge rate, nitrate leaching, annual carbon sequestration, flowering resources, and share of semi-natural-habitats. Annual biomass yield and the nitrate leaching showed the biggest differences. Unlike other research carried out in this area, the annual biomass yield was lower in LTS with AF than in NAF LTS. This was due to the different rotation length of annual crops as compared to trees, and to the annual accounting. When the AF and NAF LTS were compared over the rotation length of trees (60 to 80 years), greater total productivity tended to be achieved for AF LTS compared with NAF LTS. Similar results have been reported in previous research, where growing trees and crops together can be more productive (Sereke et al., 2015).

The groundwater recharge rate was lower in agroforestry dominated LTS, mostly due to the higher evapotranspiration by trees than by arable crops or grassland. This can also be one of the reasons for the significantly lower nitrate leaching predicted. In AF LTS modelled nitrate leaching was nearly half of that in NAF LTS, pointing to a clear ES benefit in terms of reduced nutrient emissions to the environment. This echoes similar findings by e.g Nair et al. (2007) and Jose (2009), who showed that agroforestry systems can help reduce nutrient losses by 40 to 70%. López-Díaz et al. (2011) showed in greenhouse experiments that trees have a higher root density and a deeper root horizon, which led to a higher uptake of nitrate and a reduction of nitrate leaching of 38 to 85%.

Whilst annual biomass yield in NAF LTS exceeded annual yields in AF LTS, the opposite result was obtained for annual annual carbon sequestration, which was about 30% higher in landscapes with higher shares of agroforestry. This is due to the carbon sequestered on the tree biomass (above and below ground) and to higher sequestration in the soil. Our results were similar to results reported by Cardinael et al. (2015) in agroforestry plots in France, who measured an annual below ground carbon sequestration of 0.09 to 0.46 t C ha^{-1} yr^{-1} and an above ground carbon sequestration of 0.004 to 1.85 t C ha^{-1} yr^{-1} in the tree biomass. Higher carbon sequestration rates have been reported for young plantations (Nabuurs and Schelhaas, 2002).

The amount of flowering resources and the share of semi-natural Habitats were also significantly higher in agroforestry LTS, mainly because traditional cherry orchards are a rich

flower resource during spring and because they are actually mapped as semi-natural habitats in accordance with the agri-environmental objectives of Switzerland that list traditional fruit orchards as a target habitat type (BAFU and BLW, 2008). Accordingly, traditional fruit orchards can be accounted as ecological focus areas and are promoted by agri-environmental subsidies (Herzog et al., 2018, 2017).

Our research failed to account for the positive relationship between soil preservation and agroforestry systems as shown by e.g. Wezel et al. (2014). No significant difference in soil erosion between AF and NAF LTS was found. This is different to former studies, where agroforestry systems have been shown to reduce soil erosion (João H N Palma et al., 2007; Rodríguez-Ortega et al., 2014; Sánchez and McCollin, 2015). However, it is worth noting that in our LTS, topographical differences mask the soil preservation benefits associated with agroforestry systems since the cherry systems occurred on steeper terrain than arable uses (20% slope for AF LTS as compared to 9 % for NAF LTS).

Directly interlinked to the findings on biomass stock was the carbon stock indicator, although – in addition to the carbon stored by the trees – it also comprises the carbon storage potential of the soil. Still, the overall differences between AF and NAF landscapes were relatively small. This was mainly due to the composition of the LTS, both of which included substantial areas of forest, which provides the greatest sequestration benefit. Nonetheless, the use of agroforestry systems would provide some carbon sequestration benefits whilst allowing food production to continue.

While previous studies assessing pollination services (e.g. Kennedy et al. 2013; Schüepp et al. 2013) highlighted the importance of woody elements in landscapes, we did not find significant differences between AF and NAF LTS. The size of the LTS (1 x 1km) did not allow to detect any effect that the higher availability of flowering resources in AF LTS this could have on the pollination service, because the moving corridors of the pollinators were larger than the LTS themselves. Three sizes of moving corridors were assessed, but differences between AF and NAF LTS only became significant at the 100 m level. This suggests that pollinators can subsist in both landscape types, but that fitness, resilience, and resistance of each individual might be greater in the AF LTS.

The indicator SIDI was found to be statistically similar in both AF and NAF LTS, because the index is largely driven by the number of habitat types. In fact, it was slightly greater in NAF LTS, because different crop types were counted as different habitat types. The ToSNH indicator

was similar across all LTS, although a wider range of semi-natural habitat types (6 - 40) occurred in the AF LTS. Former studies suggest that biodiversity might be better supported in AF LTS than in NAF LTS. Birrer et al. (2007), and Bailey et al. (2010) have shown that fruit orchard landscapes in temperate Europe have relatively high species richness as well as specialised species such as orchard birds.

Given that our findings are based on a limited number of LTSs and field data, the results from the analysis should be treated with considerable caution. However, agricultural landscapes tended to provide a higher amount of provisioning services, while in agroforestry landscapes regulating ES were better represented. Those conclusions are supported by similar investigations of ES provided by agroforestry systems in other parts of Europe (Kay et al., 2018b).

2.5 Conclusion

Our study explored the ecosystem services supply from agroforestry systems from a landscape perspective by developing a spatially explicit model. Fifteen indicators were chosen to represent six ES (biomass production, groundwater recharge, nutrient retention, soil preservation, carbon storage, habitat and gene pool protection). To our knowledge, this is the first attempt to comprehensively quantify ecosystem services with a semi-quantitative approach at the landscape scale through a combination of field investigations and modelling. The approach thus goes beyond expert evaluations and modelling results. The approach is limited by the availability of spatial data (notably high-resolution soil maps) and by the state of the art of modelling, which reflects our current understanding of the relevant processes. However, this approach provides an example for spatially explicit quantification of provisioning and regulating ES and is suitable for comparing different land use scenarii at a landscape scale.

The model was applied to a traditional agroforestry system, a cherry orchard landscape in Switzerland. We found that the provisioning ES was higher in LTS dominated by arable land use, while the regulating ES were higher in LTS with agroforestry. The modelling approach is thus capable to capture such differences at the landscape scale. It can be tested in other regions and for other agroforestry systems. It could also be adapted for applications outside the specific agroforestry context.

Chapter 3

Spatial similarities between European agroforestry systems and ecosystem services at the landscape scale

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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 study regions from three European biogeographical regions: Mediterranean (montado and dehesa), Continental (orchards and wooded pasture) and Atlantic agroforesty 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 results for 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,

3.1 Introduction

Around forty percent of the European land area is used for agriculture (Eurostat, 2013). Farmers cultivate the land to ensure food, fodder, energy and material supply, and by doing so, they shape the rural landscape (van der Zanden et al., 2016). Structural changes in agriculture, due to mechanisation and intensification of production, are thus reflected as visible changes in the landscape. Larger fields and farms as well as the removal of landscape elements such as trees, hedgerows, or wet areas have been some of the consequences (Biasi et al. 2016), resulting in the loss of the associated functions and environmental problems such as water pollution, erosion, and biodiversity loss (Tilman, 1999). Thus, the performance of agricultural land should not only be evaluated in relation to its production function but also in terms of demands for environmental, regulating, and aesthetic benefits from landscapes (Dale and Polasky, 2007).

The Millennium Ecosystem Services Assessment outlined the value of ecosystems and their ecosystem services (ES) into provisioning, regulating and cultural services (MEA 2003; Haines-Young & Potschin 2013) and showed how these were degrading on a global scale. Subsequently this has triggered increased efforts in measuring, quantifying and mapping ES (e.g. Maes et al. 2012a) along with assessments of synergies and trade-offs in ES (e.g. Turner et al. 2014; Mouchet et al. 2017) in order to maintain the functionality of ecosystems and their benefits to society.

Agroforestry which deliberately integrates woody elements like trees or shrubs with agricultural crops and/or livestock has been proposed as an alternative land use approach that could potentially enhance ES provision (Jose, 2009; Pimentel et al., 1992). Agroforestry systems (AF) have been identified for their high nature value and biodiversity (McNeely and Schroth, 2006; Oppermann et al., 2012) and are listed for this in the EU Habitats Directive, receiving protection under the NATURA 2000 network (European Commission, 1992). Their positive impact on all three ES pillars (provisioning, regulating and cultural, e.g. Torralba et al. 2016) and biodiversity are well studied at a local scale in wooded pastures (Moreno et al., 2016b) and fruit orchards (e.g. Bailey et al. 2010), but little research exists on the benefits of agroforestry systems at the pan-European scale.

This paper therefore explores the potential of traditional temperate agroforestry systems to provide provisioning and regulating ES and investigates their spatial impact at the landscape scale. The cultural ES provision is presented by Fagerholm et al. (2016). We conducted case studies in three European biogeographical regions (Mediterranean, Continental and Atlantic). The study aimed to answer two specific research questions: (1) Do agroforestry practices enhance landscape in comparison to agricultural land by providing additional regulating ES?

(2) Are these effects similar in different regions even though the specific types of agroforestry are different? In order to answer these two questions, we identified a set of case study areas in our selected biogeographical regions, modelled the provision of ES for each agroforestry system in those areas, and then aggregated the findings to make our assessment across all the case studies.

3.2 Data and Methods

Six traditional European agroforestry landscapes (extent > 50 km^2) in Mediterranean, Continental, and Atlantic regions were selected. In each region, four to seven adjacent municipalities were chosen and land use was broadly classified into agriculture (non-agroforestry) and agroforestry based on regional land use classification. In each of these two categories, four landscape test sites (LTS) of 1 km x 1 km each were selected randomly. A field protocol was used to map the habitats and the AF trees or AF hedgerows via a combination of aerial photograph interpretation and fieldwork in all LTS in a uniform manner. Field data were digitised and intersected with AF elements to generate habitat maps that allowed to undertake spatial ES assessment.

3.2.1 Case study regions

The selected case study regions with typical agroforestry systems were: (1) montado in Portugal, (2) dehesa in Spain, (3) cherry orchards and (4) wooded pastures in Switzerland, (5) chestnut soutos in Spain and (6) hedgerow agroforestry landscapes in the United Kingdom. The systems differ in character, management and objectives. Figure 10 shows the location of the regions, the composition and pictures of the LTS.

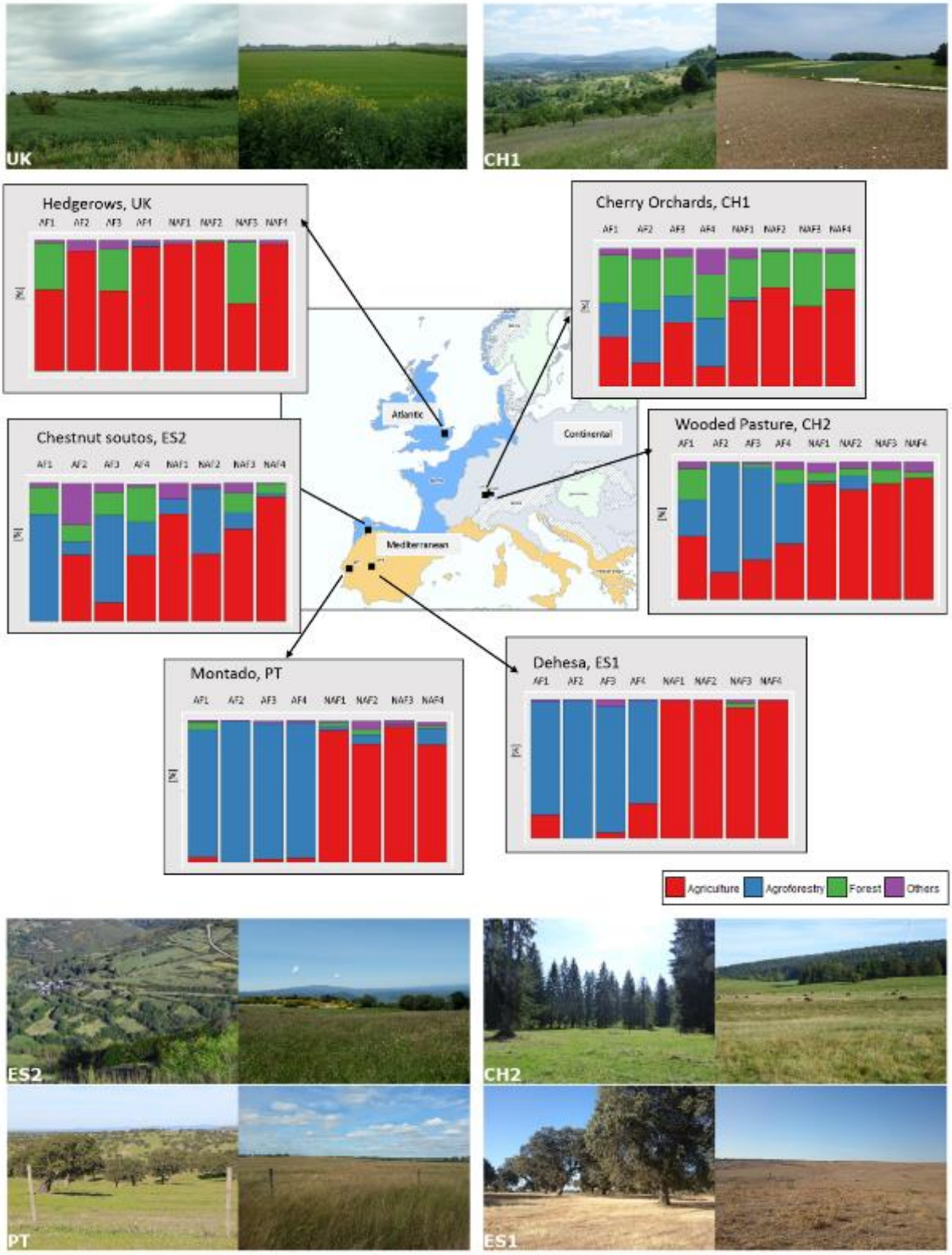


Figure 10: Location of the case study region, habitat composition and pictures of agroforestry (AF) and non-agroforestry (NAF) landscape test sites (LTS).

(1) Montados occupy an area of 736,775 hectares in Portugal (AFN, 2010) and are characterized by low density trees (25-50 trees ha^{-1}) combined with agriculture or pastoral activities (Pereira

and Tomé, 2004). The main tree species are cork oak (*Quercus suber* L.) and/or holm oak (*Quercus rotundifolia* Lam.). Depending on the main tree species present, two different types of "montados" exist: 1) Cork oak montado where cork extraction is dominant, and 2) Holm oak montado where livestock (mainly cattle or sheep) are grazed during spring and Iberian pigs feed on acorns in autumn (Gaspar et al., 2007). The habitat mapping was done in Montemor-o-Novo, located in Central Portugal.

(2) Dehesas are very similar to holm oak montados, with *Quercus ilex L*. In Spain dehesas occupy around 3.5 million hectares of land (Plieninger et al., 2015a) and has a random pattern of around 25 trees ha^{-1} where permanent grassland provides fodder in the form of acorns and grass for animal production. In addition to this, the timber and many other non-timber products are used (Fagerholm et al., 2016). The LTS selected for the ES assessment were located in Trujillo, in the southern Spanish region of Extremadura.

(3) The cherry (*Prunus avium* L.) orchards are located in the north-western part of Switzerland. Traditional fruit orchards are widespread in central Europe (approximately 1 million hectares, (Herzog 1998a)) and were mainly established for subsistence and commercial fruit production. The cherry orchards in the Cantons of Solothurn and Basel-Landschaft usually consist in 50 – 80 trees ha^{-1} of mixed age on permanent grassland that is grazed with cattle and occasionally mown.

(4) The spruce (*Picea abies* L.) dominated wooded pastures, are located in the Jura mountains in western Switzerland, covering about 50,000 hectares (Herzog, 1998b). Wood pastures are common in mountain areas and typically consist of dense and sparse woodland in a mosaic pattern (Buttler et al., 2009). The trees produce timber and fodder, typically for free ranging cattle and horses. The case study site was located around Saignelégier in the Canton Jura.

(5) Chestnut (*Castanaea sativa* Miller) soutos are a traditional land use system in north-western Iberia (Nati et al., 2016). They consist of ancient valuable trees (400 years old), are protected by the NATURA 2000 habitat network and occupy more than 350,000 hectares of land in Galicia and about 40,000 ha in Portugal. The system produces chestnut, fruit, and timber. In addition, it is known for mushroom production and in some areas grazed with pigs (Rigueiro-Rodríguez et al., 2014). The case study site was located in the western mountains of Lugo province in Galicia (Spain).

(6) The hedgerows landscape in eastern England covers around 551,000 hectares of land and is widely spread in the UK (den Herder et al. 2017). The case study region near Thetford, in the Breckland district of Norfolk, consists of cereal crops surrounded by hedgerows. These contain several species of broadleaf trees and shrubs that were traditionally used for firewood. In

addition to their use for marking field boundaries (living fence), they are used as a wind-break to reduce soil erosion by wind.

3.2.2 Indicator assessment

For each LTS we evaluated seven ES indicators; namely biomass yield and groundwater recharge rate as provisioning ES and the regulating services nitrate leaching, carbon sequestration, soil erosion, and biodiversity divided into pollination and habitat richness. The selection follows the Common International Classification of Ecosystem Services (CICES) classification (Haines-Young and Potschin, 2013) with focus on relevant indicators in agriculture and agroforestry systems. The indicators, methods and data sources are summarised in Table 4.

Table 4: Ecosystem services indicators, methods and references.

CICES Section - Division		**ES indicator**	**Model**	**Unit**	**References**
Provisioning	Material	Biomass yield	EcoYield-SAFE	t dry matter ha^{-1} yr^{-1} t dry matter ha^{-1}	Palma et al, submitted.; van der Werf et al. 2007
	Water	Groundwater recharge rate	Water balance	mm	Allen et al. 1998; Hürdler et al. 2015
Regulating and maintenance	Nutrient retention	Nitrate leaching	MODIFFUS 3.0	kg N ha^{-1} yr^{-1}	Hürdler et al. 2015
	Soil preservation	Erosion	RUSLE	t soil ha^{-1} yr^{-1}	Renard et al. 1997; Panagos et al. 2015
	Climate regulation	Carbon sequestration	EcoYield-SAFE, Yasso07	t C ha^{-1} yr^{-1} t C ha^{-1}	Liski et al. 2005; Palma et al. submitted
	Pollination	Pollination	Lonsdorf	%	Lonsdorf et al. 2009
	Gene pool protection	Habitat richness	SIDI, SoSNH, HD	Unitless	Bailey et al. 2007; Billeter et al. 2008

Indicators were calculated using spatial ES assessment models based on the habitat maps in combination with climate (online climate tool CliPic, (Palma, 2017), soil (European Soil Database (ESDB)) and topographical information (International Centre for Tropical Agriculture (CIAT), digital elevation model (DEM) by Reuter et al. (2007) and Jarvis et al. (2008)) for each case study region.

The estimations of AF trees biomass production, crop yields and carbon sequestered (divided into annual use e.g. cereals, fruits, prunings, timber and total stock) by the systems' above and

below ground biomass were provided using the EcoYield-SAFE model, a process-based agroforestry growth model that was calibrated for the assessed systems (Palma et al., submitted). In the hedgerow agroforestry landscape in the UK, observed data from farms were utilised. The average yield for cropland production came from FAO (2017).

The groundwater recharge rate was assessed using the water balance equation, which links precipitation (P), evapotranspiration (E), surface runoff (R) and the belowground water exchange (ΔS). The latter is the sum of storage change in the soil (ΔS_{Soil}) and the ground water recharge ($\Delta S_{Groundwater\ recharge}$) (Equations 1, 2).

$$P = E + R + \Delta S \qquad with\ \ \Delta S = (\Delta S_{Soil} + \Delta S_{Groundwater\ recharge}) \qquad \textit{(Equations 1, 2)}$$

Precipitation was based on climate data for each case study region from the online climate tool CliPick (Palma, 2017). Evapotranspiration was calculated by the FAO Penman-Monteith equation (Allen et al., 1998) and the MODIFFUS 3.0 methodology (Hürdler et al., 2015) was applied to assess the surface runoff. The groundwater recharge rate (GWRR) involves the amount of rainfall that percolates into the groundwater (Equation 3).

$$GWRR = \frac{\Delta S_{Groundwater\ recharge}}{P} * 100 \qquad \textit{(Equation 3)}$$

In particular, DEM and soil information obtained from Panagos et al. (2012); Hiederer (2013); Ballabio et al. (2016); Makó et al. (2017) were used.

The assessment of nitrate leaching was based on the water cycle modelling and by deploying the MODIFFUS 3.0 method (Hürdler et al., 2015), an empirical model for nitrate and phosphorus losses. Herein leaching values for each land cover class weighted by factors for soil characteristics, fertilizer application, and drainage were set.

The RUSLE equation (Renard et al., 1997) was applied to assess soil loss by water. Herein the rainfall-runoff erosivity factor (R) is multiplied by the soil erodibility factor (K), the slope length factor (L), the slope steepness factor (S), the cover management factor (C) and the support practice factor (P). These results in the average soil loss (A) (Equation 4).

$$A = R * K * L * S * C * P \qquad \textit{(Equation 4)}$$

The spatial data were provided from the European Soil Database (ESDB) (in particular Panagos et al., 2014, Panagos et al., 2015, Panagos et al., 2016).

Carbon sequestration was estimated as the sum of above and below ground crop and tree biomass, based on EcoYield-SAFE and in addition the soil organic carbon (SOC), modelled in YASSO0.7 (Liski et al., 2005). The YASSO model was primarily developed for forest stands,

focusing on the decomposition of biomass fractions and their effects on soil carbon. The carbon assessment was divided into annual sequestration rate and total carbon stock.

The biodiversity assessment was divided into functions and capacities of nature represented by pollination and habitat richness and diversity. Lonsdorf et al. (2009) equations were spatially applied for evaluating the pollination potential for cavity and ground nesting species for 100 and 350 m flight and foraging distances. As a pre-requisite, flowering and nesting facilities for wild pollinators were recorded during the habitat mapping (except for the UK case study region). Landscape metrics, computed from the habitat maps of the LTS, were used as proxies for habitat richness (Billeter et al., 2008), particularly the Simpson diversity index (SIDI), the share of semi-natural habitat (SoSNH) and the number of semi-natural habitat types (HD).

The analysis of ES was conducted on two spatial levels. Firstly, the analysis was done at regional level comparing agroforestry and non-agroforestry LTS of each case study region separately. Secondly the results were aggregated at a landscape level including all LTS. All results were statically tested using t-tests and linear regressions in R (R Development Core Team 2013). The spatial analysis was performed in ArcGIS10.4 (ESRI 2016) and SAGA GIS (Conrad et al., 2015). The methods were described in detail by Kay et al. (submitted).

3.3 Results

Examples of the LTS habitat maps are shown in Figure 12. The range of results, separately per ES indicator, obtained from the model are summarised in Figure 12. Herein the spatially explicit results are aggregated to case study level, divided into agroforestry (AF) and non-agroforestry (NAF) LTS and arranged into Mediterranean, Continental and Atlantic regions. The analysis was done (i) for each case study and (ii) aggregated across all case study regions.

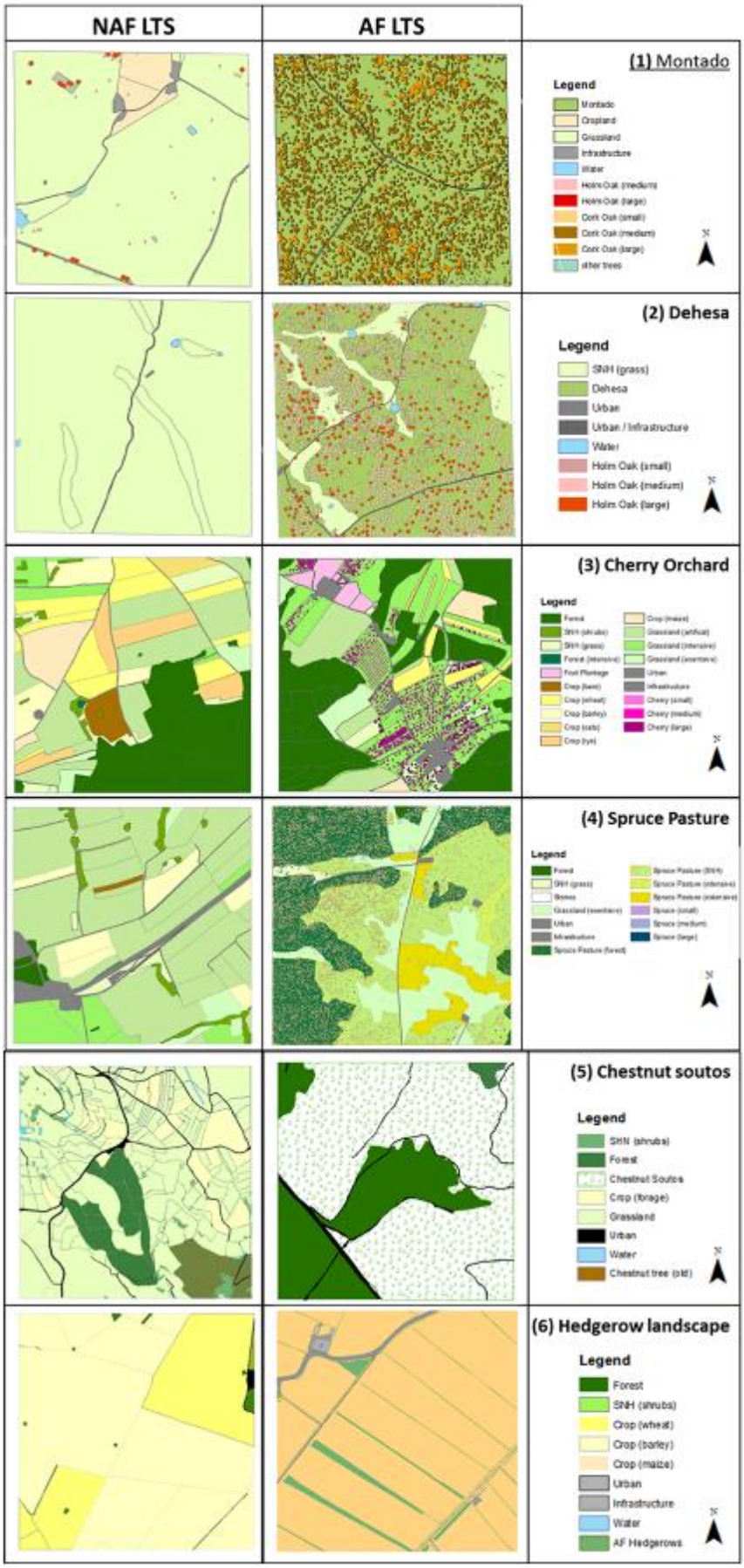


Figure 11: Examples of habitat maps of an agroforestry (AF) and a non-agroforestry (NAF) landscape test site (LTS) for each case study region.

3.3.1 Biomass

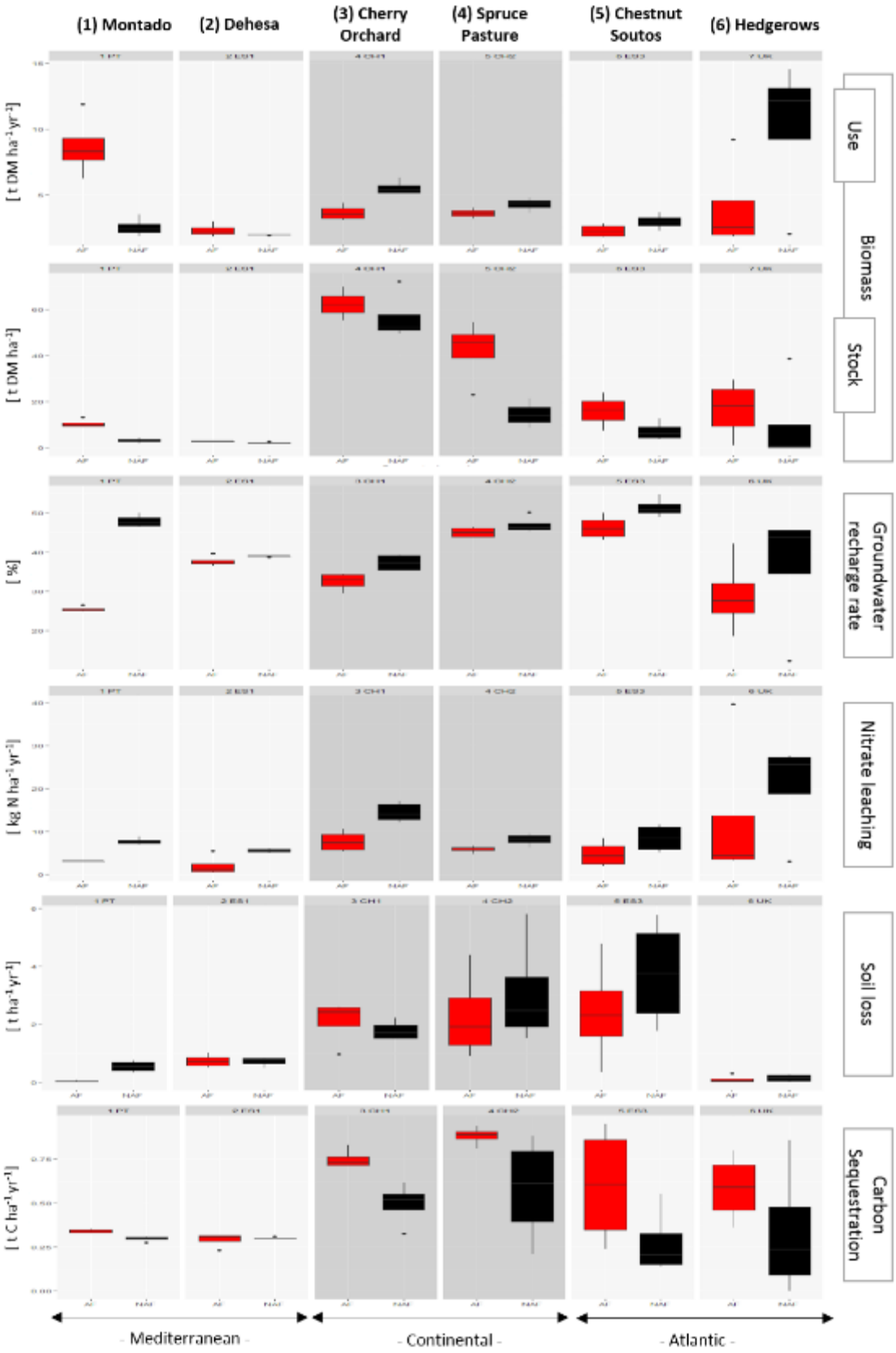
The annual use of biomass, mainly the crop yield plus tree prunings, ranged from 1.7 up to 14.5 t ha^{-1} yr^{-1} with an average of 4.3 t ha^{-1} yr^{-1}. In most regions, agricultural NAF LTS showed higher annual yields than AF landscapes. Exceptions were the Mediterranean systems, where the agroforestry LTS produced higher yields. Statistically validated differences between NAF and AF plots were found for montado, cherry orchards and spruce pasture (Table 5 and Figure 12). Over all regions, the variation between AF and NAF LTS was not statistically significant. For the total stock value at any one time (t DM ha^{-1}), which represents mainly the total volumen of timber, the trends were reversed. With 25 t ha^{-1}, AF landscapes had higher average biomass stocks than NAF (15.6 t ha^{-1}). The outcomes varied between 0.1 t ha^{-1} to 72 t ha^{-1}. The overall comparison showed no significant difference between AF and NAF, while in montado, dehesa and spruce pasture significant variations were found.

3.3.2 Groundwater recharge

The groundwater recharge rate varied between 18 and 54 % of the annual precipitation. The lowest values were obtained in agroforestry landscapes in the United Kingdom, while the highest values were in non-agroforestry LTS in Galicia and Portugal. The evapotranspiration was always higher in agroforestry areas. The recharge rate in AF LTS ranged between 28.7 % and 46.4 % with an average of 36.9 %. In NAF LTS the range was higher: between 35.3 % to 54.9 %, with an average of 43.6 %. These differences were statistically significant across all regions ($p<0.01$).

3.3.3 Nitrate leaching

Values for nitrate leaching were very low, especially in southern Europe. They ranged between nearly 0 up to 37 kg N ha^{-1} yr^{-1}. AF LTS tended to leach less nitrate than NAF LTS; in average 5.2 kg N ha^{-1} yr^{-1} in AF as compared to 9.9 kg N ha^{-1} yr^{-1} in NAF. These overall differences between land cover classes were significant ($p<0.05$). Within the regions, cherry orchards and spruce pasture in Switzerland, dehesa and montado showed statistically verifiable variations between agroforestry and non-agroforestry test sites (Table 5).


(1) Montado
(2) Dehesa
(3) Cherry Orchard
(4) Spruce Pasture
(5) Chestnut Soutos
(6) Hedgerows
Use
Biomass
Stock
Groundwater recharge rate
Nitrate leaching
Soil loss
Carbon Sequestration
[t DM ha-1 yr-1]
[t DM ha-1]
[%]
[kg N ha-1 yr-1]
[t ha-1 yr-1]
[t C ha-1 yr-1]
- Mediterranean -
- Continental -
- Atlantic -

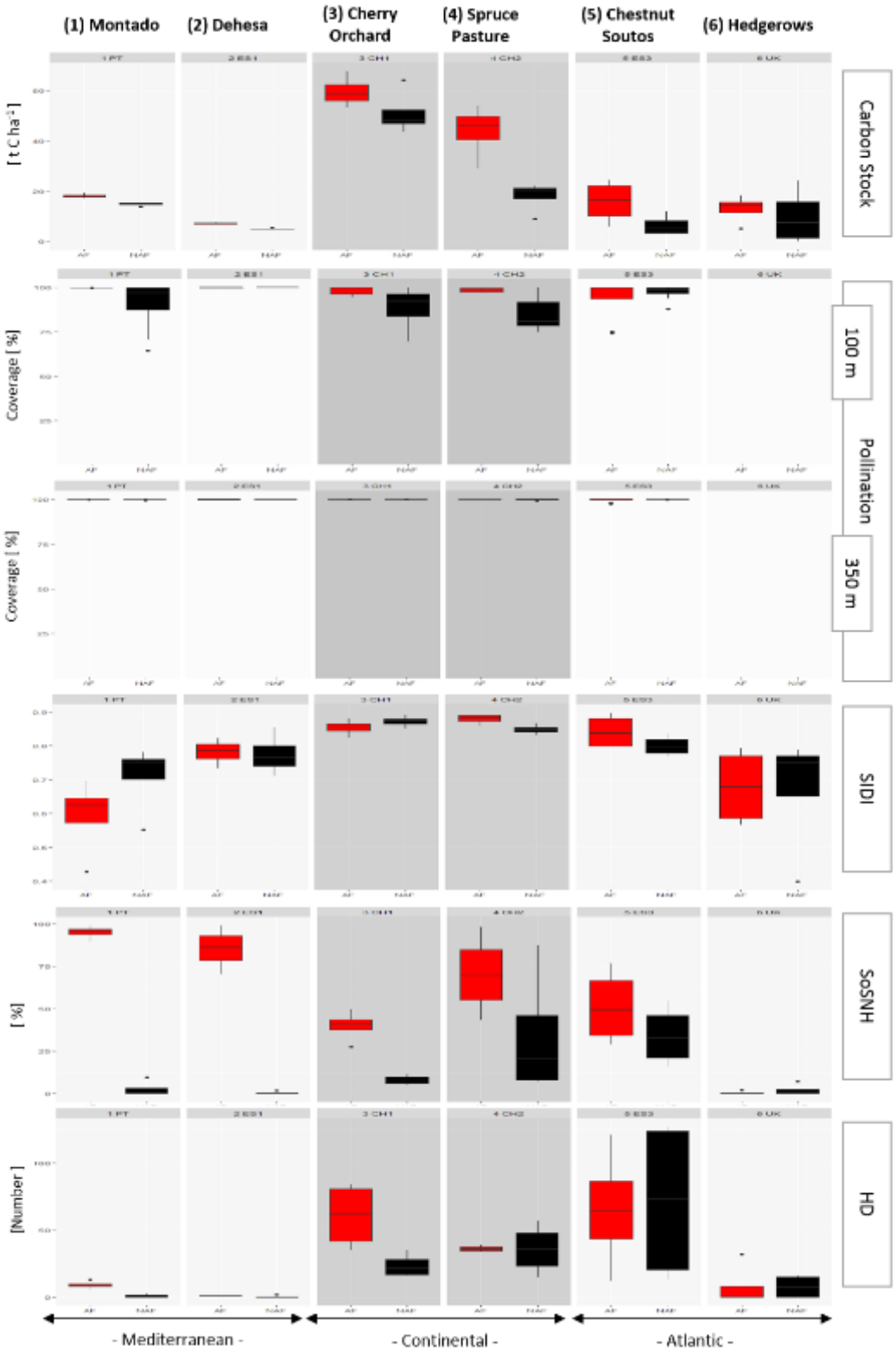


Figure 12: Summary of ES assessment grouped into agroforestry (AF - red) and non-agroforestry (NAF - black) landscape test sites for each case study region clustered into Mediterranean, Continental and Atlantic regions. Pollination services could not be evaluated for the UK. The bar graphs indicate mean values (horizontal line), standard deviation (upper and lower limits of boxes), range of values (lines) and outliers (points) [SIDI: Simpson's diversity index, SoSNH: share of semi-natural habitat, HD: Habitat Diversity]

3.3.4 Soil loss

The indicator of soil loss showed strong variations within and across regions. The average loss was 1.39 t ha^{-1} yr^{-1} in AF, covering a range of 0.01 to 4.7 t ha^{-1} yr^{-1}, and 1.59 t ha^{-1} yr^{-1} in NAF (0.04 - 5.80 t ha^{-1} yr^{-1}). No significant differences were found among AF and NAF LTS across all regions and within case study regions. Because soil loss and topography are closely interlinked, we tested soil loss against slope. Standard multiple linear regression models were used to relate AF and NAF LTS (Figure 13), p-values for slope were statistically significant ($p < 0.01$) and showed a reducing effect of AF on soil loss.

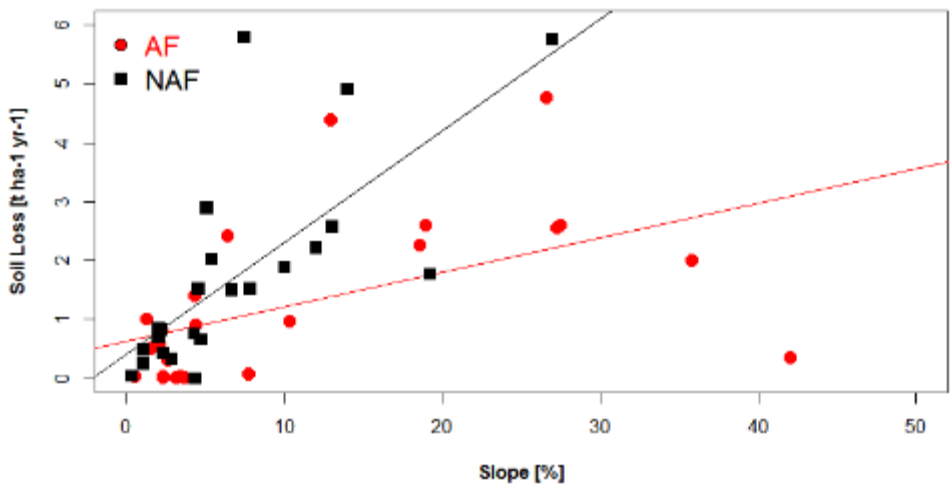


Figure 13: Erosion assessment grouped into agroforestry (AF, red) and non-agroforestry (NAF, black) landscape test sites as a function of the slope. [p-value: 1.395e-05, Adjusted R2: 0.394]

3.3.5 Carbon sequestration

The carbon assessment was divided into an annual carbon sequestration rate and the total carbon stock. The model results varied strongly within and across case study regions. In the overall trend agroforestry landscapes sequestered on average 0.57 t C ha^{-1} yr^{-1}, while in NAF the value was around 0.37 t C ha^{-1} yr^{-1} ($p<0.01$). The lowest average C sequestration rate was in cropland dominated landscapes in the UK and the highest in an agroforestry LTS in Switzerland. Results showing significant differences were found in the montado and the cherry orchards.

The model outcomes for carbon stock were similar to the carbon sequestration rate: in all case study regions, the agroforestry landscapes had a higher average amount of carbon stock compared to NAF LTS (26.2 versus 17.1 t C ha^{-1}). However, there was no overall significant difference between agroforestry and non-agroforestry areas. Significant variation was found in montado, dehesas and spruce pasture.

3.3.6 Pollination

The model demonstrated that pollinators and their services could potentially cover the whole area of most LTS within a distance of 350 m around the nesting facilities. The only exceptions were two NAF LTS in UK, where no flowering and nesting facilities for pollinators were mapped. Significant differences were found between all AF and NAF LTS for 100 m foraging distances. For the case study regions, significant effects were found for 100 m foraging radius in montado, cherry orchards, spruce pasture and hedgerow landscapes.

3.3.7 Habitat Richness

The Simpson's diversity index assessment ranged between 0.3 and 0.89. The highest levels of diversity were recorded in the Swiss case study regions, while the lowest values were observed in a non-agroforestry LTS in the UK. None of the differences were statistically significant, though.

The variability of SoSNH was huge, with an overall trend towards a higher share of semi-natural habitats in agroforestry landscapes ($p<0.001$). In particular, this difference was statistically significant in the montado, dehesa and cherry orchard case study regions.

The indicator Total HD was also derived from mapping and showed wide-ranging values between 10 to more than 100 semi-natural habitat types per LTS. Although no correlation could be found across all case study regions, significant differences between the categories AF and NAF were revealed in the montado and the cherry orchard landscapes.

*Table 5: Summary of statistically significant differences (p-values as a result of independent 2-group t-test) between agroforestry (AF) and non-agroforestry (NAF) landscape test sites (LTS) for all Ecosystem Service indicators in each case study and across all case study sites [PT: Montado Portugal, ES1: Dehesa Spain, CH1: Cherry Orchards Switzerland, CH2: Spruce pasture Switzerland, ES2: Chestnut soutos, Spain, UK: Hedgerow agroforestry United Kingdom; *: p<0.05, **: p<0.01, ***: p<0.001, NA: Pollination services could not be evaluated for the UK; (AF): AF LTS values higher, (NAF): NAF LTS values higher]*

Ecosystem Service Indicators		**Case study regions**						**All case study regions**
		PT	**ES1**	**CH1**	**CH2**	**ES2**	**UK**	
Biomass	Use	** (AF)		** (NAF)				
	Stock	** (AF)	* (AF)		* (AF)			
Water	Recharge Rate	*** (NAF)		* (NAF)		* (NAF)		** (NAF)
Nutrient retention		** (NAF)		** (NAF)	* (NAF)			*(NAF)
Soil conservation		* (NAF)						
Carbon	Sequestration	** (AF)		* (AF)				** (AF)
	Stock	** (AF)	*** (AF)		** (AF)			
Pollination	100m cavity				* (AF)		NA	
	100m ground						NA	
	350m cavity						NA	
	350m ground						NA	
Simpson's diversity index (SIDI)					* (AF)			
Share of semi-natural habitat (SoSNH)		*** (AF)	*** (AF)	** (AF)				*** (AF)
Habitat Diversity (HD)		** (AF)						

3.4 Discussion

The results demonstrate a positive impact of agroforestry practices and systems on the supply of regulating ES at the landscape scale for all compared agroforestry systems regardless of type, region or composition. This is all the more remarkable as the agroforestry area is between 5 % in the hedgerow landscapes in the UK, where only the hedgerows are qualified as AF and around 95 % in dehesas, Spain (Figure 10). Also, most LTS included all habitat types present in the respective region, i.e. also NAF landscapes contained some agroforestry plots – although at a much lower percentage than the AF LTS. Thus, differences between ES indicator values at LTS scale are less striking than they would have been at plot scale. However, plot scale comparisons are misleading for ES that involve processes that interact spatially (e.g. erosion, pollination). Nonetheless the positive effect on regulating ES provision is directly interlinked with the amount of agroforestry in the LTS.

Nitrogen leaching mainly occurs during autumn and winter season, when the nutrient uptake of plants is limited, but also during spring caused by intensive rainfall. Approaches for reducing theses effects like using crops with higher water requirements, optimized fertilization and a

permanent, year-round land cover optimally with trees were positively examined by for example Joffre et al. (1999), Herzog et al. (2008) and López-Díaz et al. (2011). In line with those observations, considering the tree as a permanent crop, nitrate leaching in the AF LTS was systematically lower than in the NAF LTS.

García-Ruiz et al. (2015) compared erosion rates in a meta-analysis. Slope and precipitation had the highest effect on soil loss, immediately followed by land use. Our AF LTS tended to have overall higher slope percentages. As a result, there was no significant difference between AF and NAF LTS, except for the orchards, where soil loss was actually higher in the agroforestry landscapes because orchards were systematically present on steeper slopes than the non-agroforestry land uses. Only in the montado LTS, erosion was significantly reduced on AF LTS.

Due to high biomass stock and lower decomposability of tree leaves and roots (Cornwell et al., 2008), AF LTS showed higher carbon sequestration rates and higher landscape carbon stock compared to agricultural LTS. The overall high carbon storage is particularly high in the Swiss case study regions. This is mainly due to the heterogeneous landscape structure and the amount of productive forest areas in the LTS. Yet, a recent investigation in an apple intercropping system showed increased carbon soil contents already seven years after tree planting (Seitz et al., 2017). The carbon sequestration rate in spruce wooded pasture is remarkably high, following the high productivity of coniferous tree species (Bebi et al., 2013). Chestnut soutos in Atlantic climates showed slightly lower values, hence the variance was higher within the region. Interestingly, small variations were found in dehesa in comparison to montado. This may be a result of the lower tree density in dehesa and edaphoclimatic conditions, changing storage in trees that can have wide difference in carbon storage (Palma et al., 2014). Howlett et al. (2011) measured an additional soil carbon storage in oak dominated agroforestry systems of around 4 % in comparison to pasture without trees.

Zulian et al. (2013) examined pollination services at European scale. In natural reserves and areas with semi-natural habitats, the full service was determined. Agroforestry systems were qualified as semi-natural habitats and provide a high level of pollination services. Only little differences were found between AF and NAF LTS within the case study regions, mainly due to geographical proximity between the LTS and the overall complexity of the examined landscapes. Agri-environmental schemes have in general a positive impact on pollinator species richness and abundance, hence, these effects are even more strongly related to the structure and complexity of the broader landscape context (Scheper et al., 2013) .

Biodiversity needs to be evaluated at the landscape rather than at the plot scale, due to the importance of spatial interactions between habitats and species (e.g.Tscharntke et al. 2005). For the biodiversity metrics use here, differences were larger between case study regions than between AF and NAF LTS. This indicates the influence and relevance of broad landscapes contexts (> 1 km^2) in biodiversity assessments. In several case study regions, habitat diversity (SIDI) was lower in AF LTS than in NAF LTS. This is due to, for example, the lower diversity of crops in the cherry orchard landscapes and uniformally mapped AF in montados. The share of semi-natural habitats (SoSNH), on the other hand, was consistently higher in AF LTS than in NAF LTS because agroforestry systems were classified as semi-natural, in line with the European Habitats Directive (European Commission, 1992) and the European High-Nature Value categories (Oppermann et al., 2012). In the UK hedgerow landscape, however, only the area of the hedgerows were classified as SNH, which leads in total to a low SNH coverage. Comparatively fewer habitats types (HD) in montado and dehesa are again a result of their large and homogenous spatial extent (Gaspar et al., 2007) and therefore uniform mapping of these systems. Nevertheless, marginal-unmanaged habitats, even if they only occur occasionally, are crucial for biodiversity in Iberian dehesas (Moreno et al., 2016b).

Regarding provisioning ES, the results were more heterogeneous. The annual biomass use tended to be higher in NAF than in AF LTS except for montado. In this case the comparable agricultural practice was permanent grassland and in the Mediterranean climate, the presence of woody vegetation actually increases the forage availability by reducing wind speed and the water deficit in some periods of the year (Moreno and Cubera, 2008; Pardini, 2009), in addition to the acorns that also provide forage. Yield differences between the montado and dehesa case study regions could be explained by different agro-climatic conditions and tree density in the case studies (montado 50 vs. dehesa 20 trees ha^{-1}). In contrast, the biomass stock tended to be higher in AF LTS as compared to NAF LTS. This is due to the long-term biomass stored in trees. The high values in the Swiss case study regions are related to the biomass rich forest, which are part of the LTS. Variable climate conditions account for differences between the NAF landscapes in the two Swiss case study regions. While in the orchard region the focus is on cereal production, the mountain area produces mainly grass and fodder for animals. For groundwater recharge – the other provisioning ES that was evaluated, the findings were again consistent across case study regions and agroforestry systems. Vegetation cover strongly affects groundwater recharge (Campos et al., 2013) and evapotranspiration is usually higher when trees are present, due to the higher biomass stock and the increased interception of rainfall (e.g. Bellot et al. 1999; Grubinger 2015). Consequently, groundwater recharge tended to be lower in the

agroforestry landscapes across all LTS. The highest values occurred in regions with high precipitation rates, like in the chestnut soutos or the spruce pasture.

3.5 Conclusion

The spatially-explicit link between ecosystem service provision and landscape structure enables a general assessment of the contribrution of agroforestry to landscape enhancement. The multifunctionality of agroforestry systems in comparison to agricultural landscapes was reflected by reduced nitrate losses, higher carbon sequestration, reduced soil loss, higher pollination services and higher porportions of semi-natural habitats. Higher annual yields and higher groundwater recharge rates were linked to NAF areas. Whilst in traditional agroforestry landscapes the provisioning ecosystem services were lower and less biomass was leaving the system per hectare and year (with exception of Mediterranean agroforestry systems), regulating ES tended to perform better in AF landscapes.

Overall our study underlines that traditional agroforestry systems regardless of type, region and composition have a beneficial impact on the provision of regulating ecosystem services at the landscape scale. These general findings encourage to expect comparable results also for innovative agroforestry systems such as alley cropping or intercropping and grazed orchards. Against this background agroforestry systems can make a significant contribution to foster European environment policy and promote sustainable agriculture.

Chapter 4

Agroforestry is paying off - Economic evaluation of ecosystem services in European landscapes with and without agroforestry systems

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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, and 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 bioeconomic and environmental modelling. 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, ecosystem services, soil loss, external cost, groundwater recharge, nutrient loss, pollination deficit

4.1 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 (European Commission, 2013a; Somarriba, 1992). 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 evolved from a use value perspective, evolved to a monetary valuation and ended as exchange value or commodity. It ended in the question of how to cash ES in markets (Gómez-Baggethun et al., 2010; Muradian et al., 2010). 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 use values and economic performance of provisioning and regulating ES of agroforestry systems at the landscape scale. 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?

4.2 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.

4.2.1 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 6 and shown in Figure 14.

Table 6: Case study regions and the dominating agricultural (NAF, business as usual) and agroforestry (AF, alternative) system.

Alternatives	Biogeographical region	Country	Abb.	System
Agricultural NAF (Business as Usual, BAU Baseline)	Mediterranean	Portugal	PT	Open pasture
		Greece	GR	Intensive olive groves (*Olea europaea* L.)
		Spain	ES1	Open pasture
		Spain	ES2	Arable farming
	Continental	Romania	RO	Open pasture
		Switzerland	CH1	Open pasture and arable farming
		Germany	GE	Arable farming
		Switzerland	CH2	Open pasture
	Atlantic	France	FR	Mixed arable-pasture systems
		Spain	ES3	Open pasture and arable farming
		United Kingdom	UK	Arable farming

Agroforestry, AF (Alternative I)	**Mediterranean**	**Portugal**	**PT**	**Montado - Wood pasture (Cork oak, *Quercus suber* L.)**
		Greece	GR	Intercrop olive groves (*Olea europaea* L.)
		Spain	ES1	Dehesa - Wood pasture (Holm oak, *Quercus ilex* L.)
		Spain	ES2	Intercrop oak (Holm oak, *Quercus ilex* L.)
	Continental	Romania	RO	Wood pasture (Common Oak, *Quercus robur* L.)
		Switzerland	CH1	Fruit orchard (Cherry, *Prunus avium* L.)
		Germany	GE	Hedgerow landscape with arable farming (mixed species)
		Switzerland	CH2	Wood pasture (Spruce, *Picea abies* L.)
	Atlantic	France	FR	Bocage - Mixed arable-pasture systems fenced by hedgerows (mixed species)
		Spain	ES3	Chestnut soutos (*Castanaea sativa* Miller)
		United Kingdom	UK	Hedgerow landscape with arable farming (mixed species)

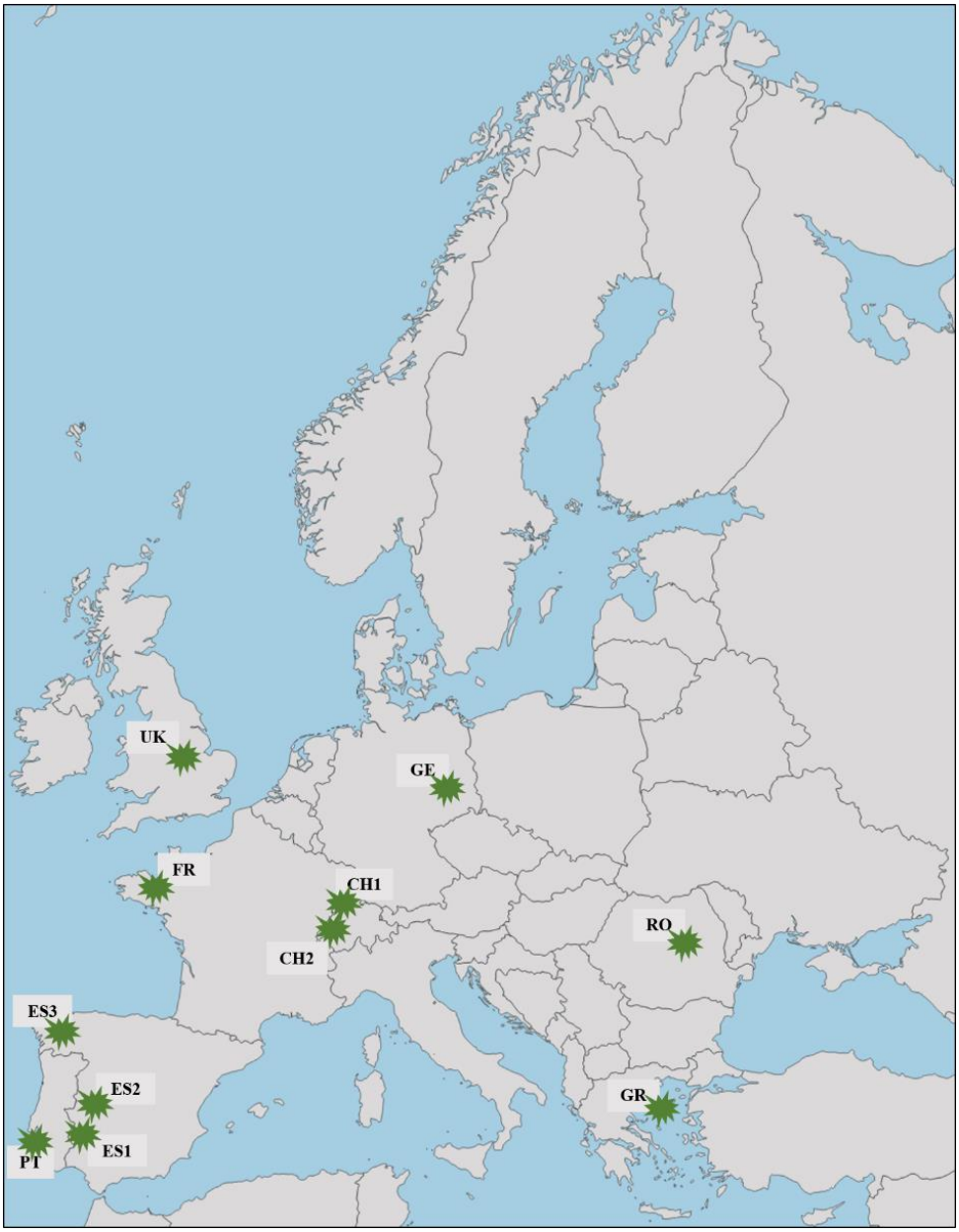

Figure 14: Location of the eleven case study regions.

4.2.2 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., n.d.) 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 96 LTS (8 LTS x 12 regions) (Kay et al., 2017; Kay et al., submitted).

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

4.2.3 Valuation and prices of market ecosystem services

Biomass production: 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, 2017c), 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. 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. 5).

$$B_{Biomass} = T_o - \frac{T_o}{i} \qquad \text{[Equation 5]}$$

$B_{Biomass}$ = 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.

4.2.4 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 (JRC Water Portal, 2017a; Roo et al., 2012).

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 (Jacobsen, 2017; OXERA, 2006).

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.

4.2.5 Summary of the net ecosystem service value

Equation 6 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 7 shows the price range and the monetary value (MV_I) of each assessed indicator.

$$V_I = Q_I * MV_I \qquad \text{[Equation 6]}$$

Table 7: Summary of prices-ranges for ecosystem services indicators and the used monetary values

Indicators		Unit	Price range	References	Used monetary value (MU_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, 2017a; Roo et al., 2012)	0.0 – 4.0 depending on country
	Carbon storage	€ t C^{-1}	1.0 – 140.0 EEX-value: 5.0	(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., 2018b; Jacobsen, 2017; OXERA, 2006)	4
	Soil loss	€ t^{-1}	0.9 – 23.0	(García de Jalón et al., 2018b; 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 7.

$$NET\ ES_{value} = S_{Biomass} + S_{Water} + S_{Carbon} - D_{Nutrient} - D_{Soil} - D_{Pollination} \qquad \text{[Equation 7]}$$

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 15 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 Annex II.

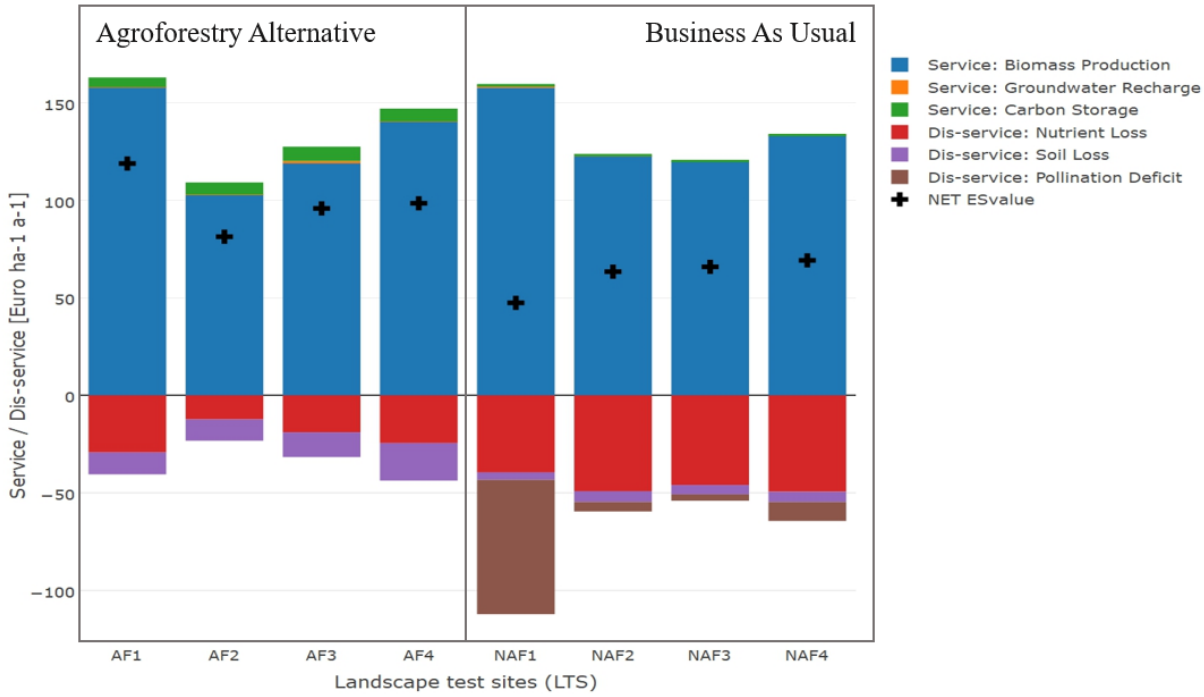

Figure 15: Visualisation of net ecosystem services value (NET ES_{value}) composition including service and dis-service indicators of biomass production, groundwater, carbon storage, nutrient loss, soil loss, and pollination deficit. Indicators were assessed in each landscape test site (LTS) and summarized 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.

4.2.6 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).

4.3 Results

4.3.1 Valuation of ecosystem services

4.3.1.1 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 16). 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 16).

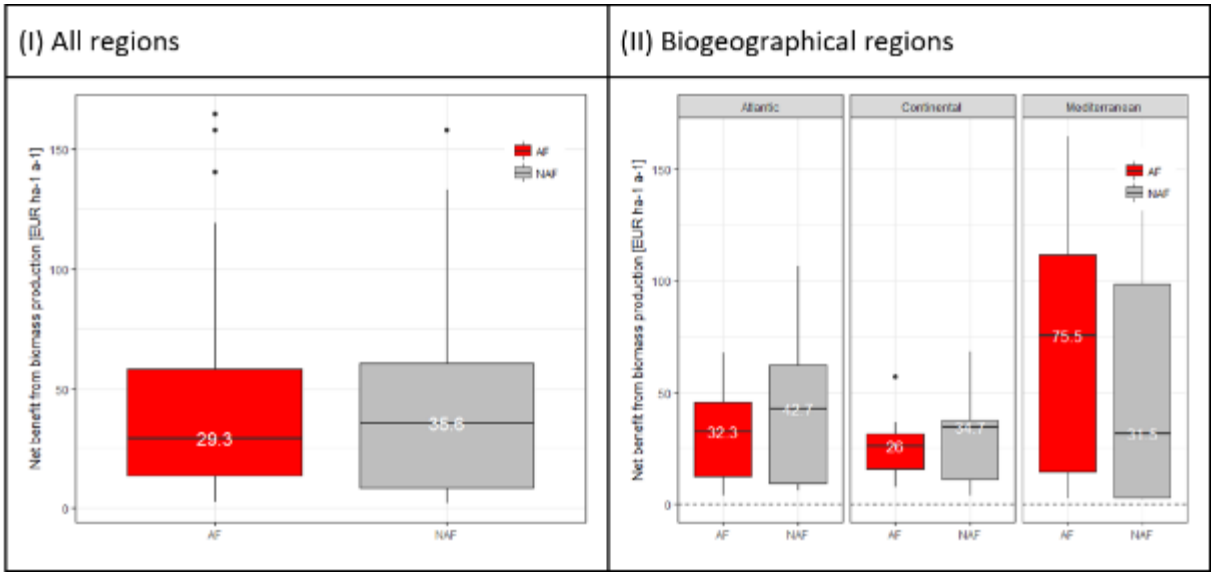

Figure 16: Average net financial benefit of biomass production [€ ha^{-1} a^{-1}] of all 12 cases study regions (I) and divided into biogeographical regions (II) based on landscape test sites [LTS] grouped by land cover categories into agroforestry (AF) and non-agroforestry (NAF, Business as Usual) sites.

4.3.1.2 Monetary valuation of individual ecosystem 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 17). 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 17 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.

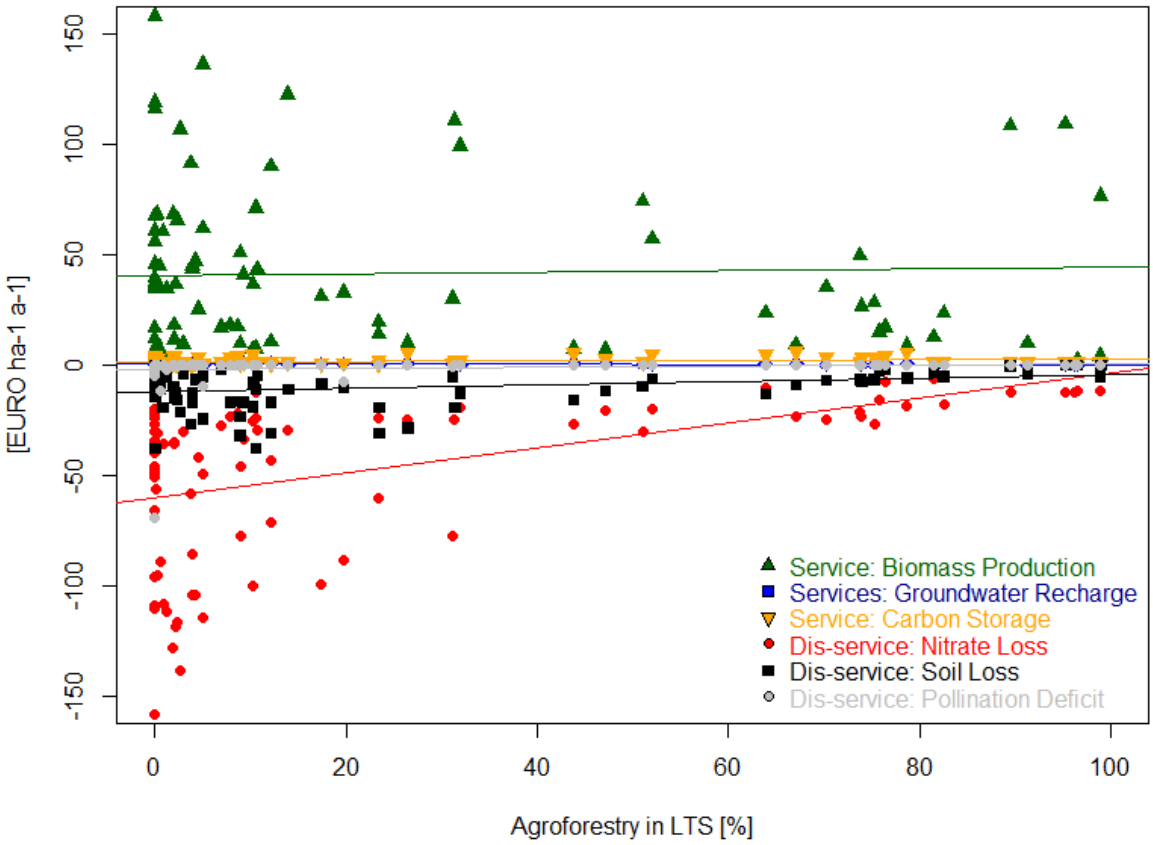

Figure 17: Monetary values [€ ha^{-1} a^{-1}] of ES indicators, depending on the percentage of agroforestry in the landscape test sites (LTS). The coloured lines are the regression line of the measurements.

4.3.1.3 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 18). 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.

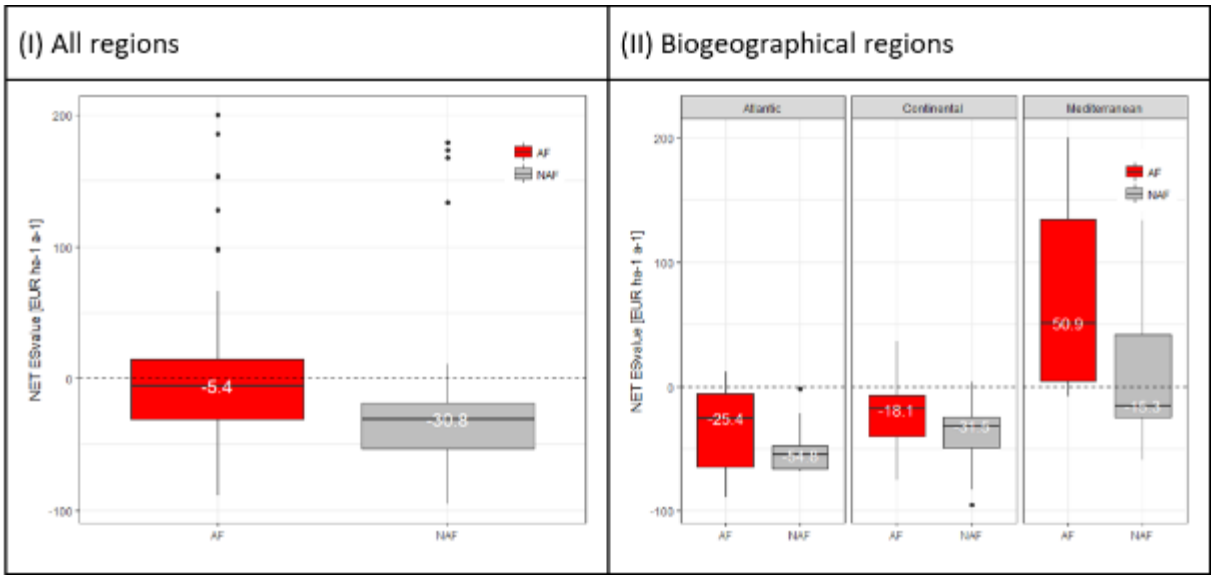

Figure 18: Net ecosystem service value in € ha^{-1} a^{-1} of all 12 cases study regions (I) and divided into biogeographical regions (II) based on landscape test sites [LTS] grouped according to dominating land cover categories into agroforestry (AF) and non-agroforestry (NAF, Business as Usual) LTS.

4.3.2 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 19). 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.

4.3.2.1 Nutrient loss

Figure 19a 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 resilience 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.

4.3.2.2 Soil loss

The soil loss assessment (Figure 19b) 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 resilience 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.

4.3.2.3 Carbon sequestration

The results for carbon sequestration (Figure 19c) 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).

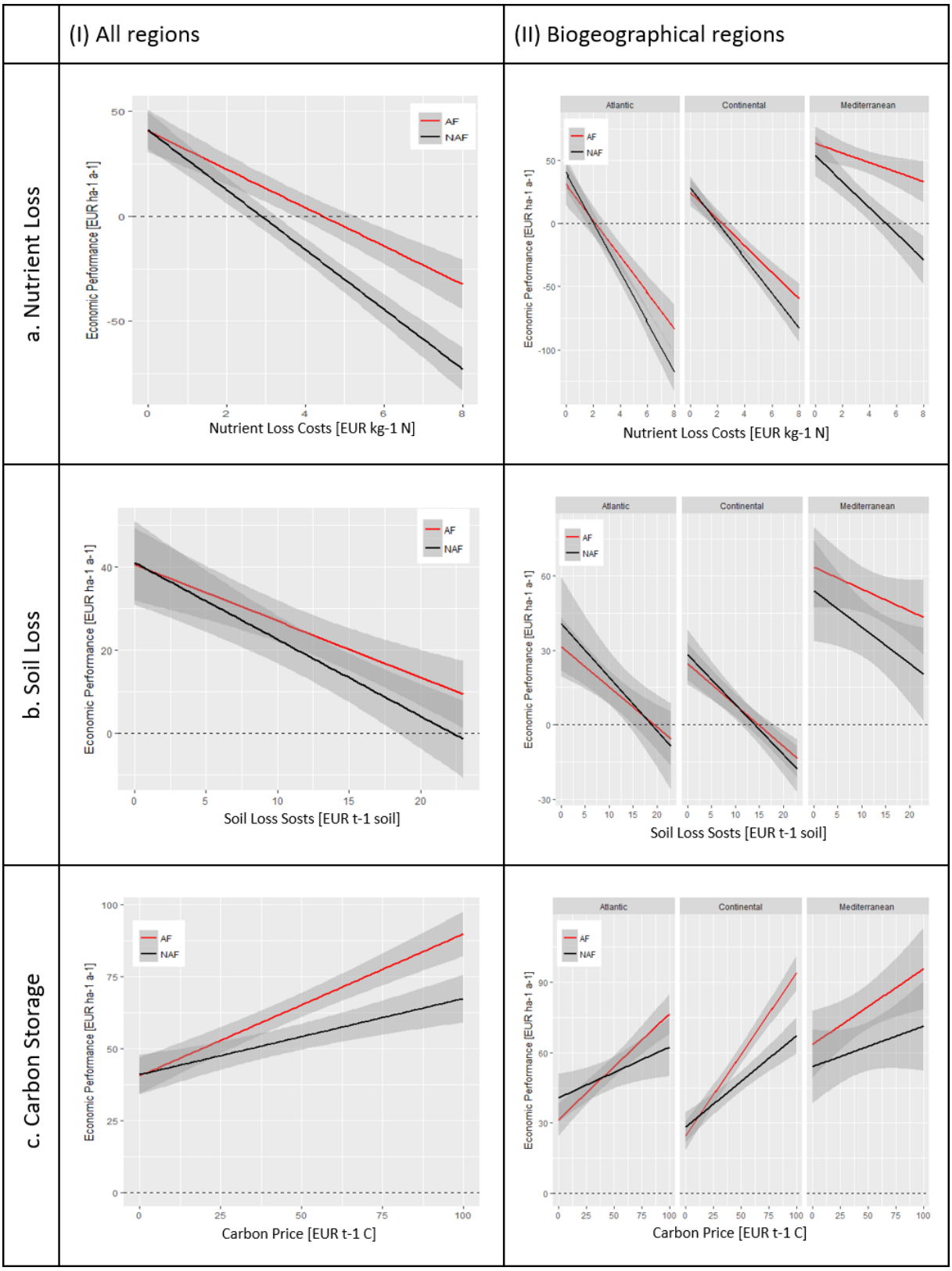

Figure 19: Economic performance of agroforestry (AF) and non-agroforestry (NAF, Business as Usual) for different ecosystem services (a) nutrient emission costs, (b) soil loss costs and (c) carbon prices together with the current sales revenues of biomass production in € ha^{-1} a^{-1} (I) over all 12 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.

4.4 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 (Breeze et al., 2014; Zulian et al., 2013). 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.

4.5 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.

Chapter 5

How much can Agroforestry contribute to Zero-Emission Agriculture in Europe? - Converting 8.9% of European farmland to agroforestry could mitigate between 1 and 43% of European agricultural greenhouse gas emissions

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Abstract

Agroforestry, relative to conventional agriculture, contributes significantly to carbon sequestration, increases a range of regulating ecosystem services, and enhances biodiversity. In a transdisciplinary approach, we combined scientific and technical knowledge to evaluate nine environmental deficits in terms of ecosystem services in European farmland and assessed the carbon storage potential of suitable agroforestry systems, proposed by regional experts. Firstly, regions with potential environmental deficits 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 deficits, loss of soil biodiversity). The maps were overlain to identify areas where several deficits accumulate. In total, 94.4% of farmlands suffer from at least one environmental deficit or more, grasslands being less affected than croplands. 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 deficits were defined as Priority Areas, where the implementation of agroforestry could be particularly effective. Secondly, 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 hedgerow on field boundaries to fast growing coppices or scattered single tree systems. Thirdly, 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, a wide range of carbon sequestration potentials was also identified, ranging from 0.09 to 7.29 t C ha^{-1} a^{-1}. While contributing to mitigate the environmental deficits, agroforestry could also sequester between 2.1 and 63.9 million tonnes C a-1 (7.78 and 234.85 million tonnes CO2eq a^{-1}) depending on the form of agroforestry implemented in the Priority Areas. This corresponds to 1.4 and 43.4 % of European agricultural greenhouse gas (GHG) emissions, respectively. This suggests that the strategic establishment of agroforestry systems could provide an effective means of meeting EU policy objectives on GHG emissions whist providing a range of other important benefits.

Keywords:

Environmental assessment, ecosystem services, deficit regions, carbon storage, climate, soil, water, biodiversity

5.1 Introduction

Increased market price volatility and the risks of changing climate are - according to the EU Agricultural Markets Briefs (September 2017) – the biggest challenges European farmers will face in near future (DG Agriculture and Rural Development, 2017). Facing the complex relationship between competitive farming and sustainable production, the current Common Agricultural Policy (CAP, the European framework for agricultural subsidies), supports farmers' income, market measures and rural development (European Commission, 2016). In spite of cross-compliance mechanism and the recently introduced greening measure that links environmental standards to subsidies, the agricultural sector is still one of the prime causes of pressure on natural resources and the environment (EEA, 2017a). To address these environmental problems, the European Commission has issued policies such as the Nitrate Directive (91/676/CEE) in 1991, the Water Framework Directive (Directive 2000/60/EC) in 2000, the Soil Thematic Strategy in 2006 (COM(2006)231) and the Biodiversity Strategy in 2010 (COM(2011) 244). Nonetheless, major environmental problems persist and are still linked to or caused by intensive agricultural production on the one hand, and by land abandonment on the other (Plieninger et al., 2016). Most recently and in line with the COP21 Paris Agreement (UNFCCC, 2015) the proposed Effort Sharing Regulation (ESR) includes a "no-debit rule" for agricultural practices (European Parliament, 2017), aiming to establish a "carbon neutral" agricultural sector, which balances greenhouse gas (GHG) emissions with an equal amount of GHG sequestration.

In this context, the future CAP for the next funding period after 2020 (CAP2020+) proposes three focal areas: a) "natural" farming, b) sustainable water management and use and c) dealing with climate change (European Commission, 2017). This will require strategies to manage the above financial and environmental risks of production, ideas to expand the agricultural product range, and a focus on sustainable farming systems with climate adaptation and mitigation functions (Wezel et al., 2014). In light of this, agroforestry, the integrated management of woody elements on croplands or grasslands (European Commission, 2013a), might play an important role in future agriculture. Agroforestry provides multiple (annual and perennial) products while simultaneously moderating critical environmental emissions and impacts on soil, water, landscapes, and biodiversity (Torralba et al., 2016). In addition, it is highlighted as one of the measures with the greatest potential for climate change mitigation and adaptation (Aertsens et al., 2013; Hart et al., 2017). For example, agroforestry can enhance the sequestration of carbon in woody biomass and in the soil of cultivated fields (mitigation) (Kim

et al., 2016), increase soil organic matter, improve water availability (adaptation to climate aridification)(Murphy, 2015), protect crops, pasture, and livestock from harsh-climate events (adaptation to global warming and increasing wind speed) (Sánchez and McCollin, 2015).

Against this background, our study aimed to evaluate the potential contribution of agroforestry towards achieving zero-GHG emissions agriculture in pursuit of the ambitious Paris Agreement COP21 and CAP targets. Using a transdisciplinary approach including scientific and practical knowledge, the study focused on three key questions: I. Where and to what extent is European agricultural land affected by (multiple) environmental deficits that could be reduced through agroforestry? II. Which regional types of agroforestry (combinations of various woody plants, crop / animal species and management practices) can be used to reduce these environmental deficits and provide multiple products? and – as an example of an ecosystem service that agroforestry can provide – III. What is the impact of the proposed systems on European climate change targets, in particular on carbon storage and GHG emissions?

5.2 Method

The study was conducted in three main phases: Firstly, the agricultural areas most seriously affected by environmental pressures ("Deficit Areas") were identified using various spatially explicit datasets on e.g. soil erosion, water pollution, and pollination deficits. Secondly, local agroforestry experts were consulted to propose suitable agroforestry practices for their regions with environmental deficits. Finally, the annual carbon storage impact of the proposed systems was identified and evaluated in the light of European and agricultural GHG emissions.

5.2.1 Priority area approach

Bearing in mind that agroforestry is only one aspect of a diversified agriculture, our focus was on agricultural areas facing combined environmental pressure, in which agroforestry can mitigate several environmental deficits. Figure 20 demonstrates the conceptual background of the Priority Area approach.

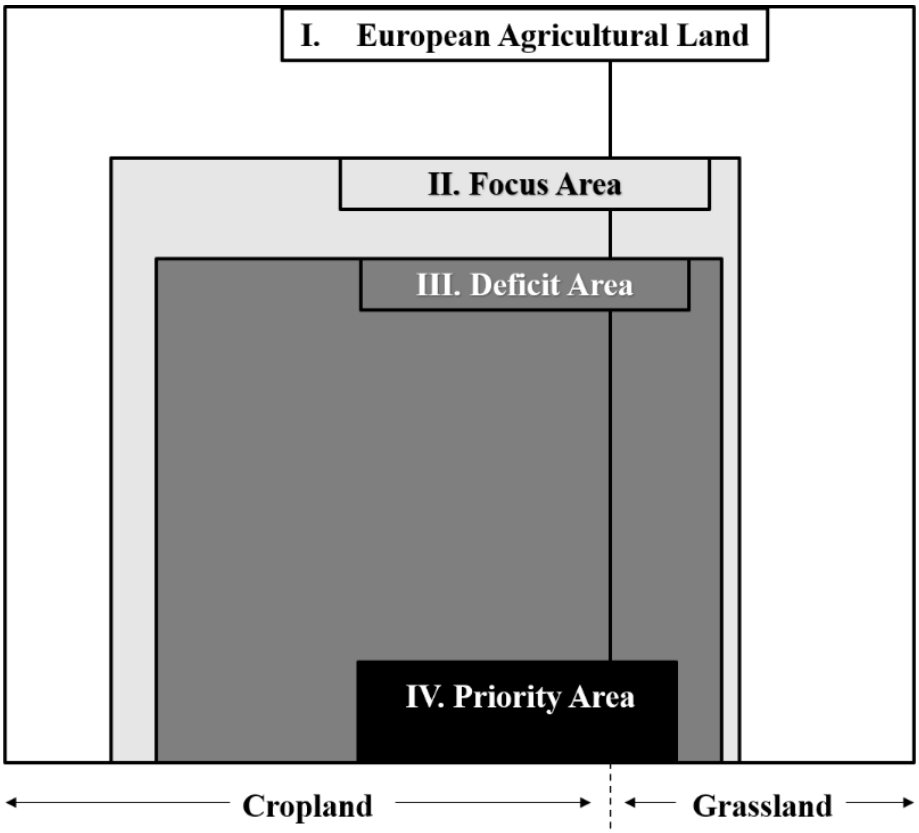

Figure 20: Conceptual approach for the spatially explicit deficit analysis. European agricultural land: Cropland and grassland. Focus Areas: European agricultural land minus nature conservation areas and existing agroforestry land. Deficit Areas: Areas where at least one ecosystem service deficit was mapped. Priority Areas: Areas where environmental deficits accumulate (four out of seven in grassland and five out of nine in cropland).

The analysis uses the Corine Land Cover 2012 (EEA, 2016) to identify the area of European arable and pasture land. From this farmland layer, the areas of high nature value such as Natura 2000 (EEA, 2015a), High Nature Value Farmland (EEA, 2015b; Paracchini et al., 2008), and the existing agroforestry areas (den Herder et al., 2017) were subtracted.

The remaining "Focus Areas" were the starting point for the deficit analysis. Environmental deficits related to: i) soil health (soil erosion by wind and water, soil organic carbon), ii) water quality (water pollution by nitrates, salinization by irrigation), iii) climate change (rising temperature), and iv) biodiversity (pollination and pest control deficits, reduced soil biodiversity) were derived. Individual deficit maps were spatially aggregated and combined into the "Deficit Areas" map showing all regions where one or several environmental deficits occur. To identify the "Priority Areas" for intervention, the sum of deficits per spatial unit (pixel size = 100m x 100m) was expressed as an accumulation map or a "heatmap of environmental deficits".

5.2.2 Deficit area analysis

5.2.2.1 Soil health deficits

The European water erosion map (Panagos et al., 2015c) and the Swiss soil erosion risk map (Prasuhn et al., 2013) together with the European wind erosion map (Borrelli et al., 2017) were used to locate areas with potentially critical loads of soil losses. According to Panagos et al. (2015) a critical threshold is reached if the soil loss is more than 5 t soil ha^{-1} a^{-1}. The analysis of potential wind erosion was limited to arable land.

The Soil Organic Carbon (SOC) saturation capacity provided at European level by Lugato et al. (2014a, 2014b) expresses the ratio between actual and potential SOC stocks. Regions with a ratio of less than 0.5 were identified as Deficit Areas, meaning that these soils contain less than half of their SOC storage potential.

5.2.2.2 Water quality deficits

Irrigated fields regardless of whether they were grassland or cropland were included in the deficit analysis. Irrigation maps were provided by the JRC Water Portal (2017) and the Farm Structure Survey (FSS) (Eurostat, 2017a) and expressed the proportion of irrigated land on the total agricultural area. Regions with more than 25% of the agricultural area under irrigation were included as Deficit Area.

The nitrogen surplus, which can lead to both high levels of nitrate leaching and denitrification to gaseous nitrous oxide, was assessed for the European Union using the CAPRI model by Leip et al. (2014). For Switzerland data were obtained from modelled accumulated nitrogen losses (BAFU, 2015). According to the German Ministry of Environment (BMUB, 2017), there is a critical load if the annual nitrogen surplus exceeds 70 kg N ha^{-1} a^{-1} and this threshold was used to identify the area of nitrogen surplus.

5.2.2.3 Deficits related to changing climate

Annual mean temperatures from the current climate (1970-2000 WorldClim; Hijmans et al. 2005) and the forecast for 2050 (HadGEM2-ES) were used to derive the predicted regional temperature increase up to 2050. In Paris, the 21st Conference of the Parties (COP21 Paris Agreement, UNFCCC, 2015) agreed to keep global temperature increase to within 2ºC by 2100. According to Hart et al. (2012), agroforestry systems remain robust within an average temperature increase of 4°C. Therefore, all areas with a predicted increase of temperature of more than 2°C and less than 4°C were qualified as deficit areas with potential for agroforestry.

5.2.2.4 Biodiversity deficits

Soil fauna, microorganisms and biological functions derived from the spatial analysis by Orgiazzi et al. (2016) were used to assess soil biodiversity. The areas identified with “high” and “moderate-high” levels of risk were defined as deficit areas.

The pollination assessment was based on the analysis of landscape suitability to support pollinators by Rega et al. (2017). Areas with “very low” and “low” suitability were defined as deficit areas.

The pest control index (Rega et al., 2018) was used as input for the assessment of regions with a potential deficit in natural pest control. Again, areas with the index classes “very low” and “low” were combined and defined as deficit areas. This analysis was limited to cropland.

Table 8 summarizes all spatial datasets used, their source, resolution and the thresholds that were applied.

Table 8: Spatial datasets with their respective characteristics and the threshold applied to define Deficit Areas (EU28: Austria, Belgium, Bulgaria, Croatia, Cyprus, Czech Republic, Denmark, Estonia, Finland, France, Germany, Greece, Hungary, Italy, Ireland, Latvia, Lithuania, Luxembourg, Malta, the Netherlands, Poland, Portugal, Romania, Spain, Slovakia, Slovenia, Sweden, and the United Kingdom; EU 27: is without Croatia; CH = value for Switzerland)

Indicator		Source	Countries	Resolution	Threshold
Focus Area	CORINE - Agricultural Land	EEA 2016	all Europe	250 m	
	Agroforestry area	den Herder et al. 2017	EU 28, CH	100 m	
	High Nature Value Farmland	EEA 2015b; Paracchini et al. 2008	all Europe (without Greece)	100 m	
	Natura 2000, Ramsar Areas	EEA 2015a	EU 28, CH		
Soil Deficit Regions	Soil erosion by water	Panagos et al. 2015; Prasuhn et al. 2013	EU 28, CH	100 m	> 5 t soil ha^{-1} a^{-1} (Panagos et al. 2015)
	Soil erosion by wind	Borrelli et al. 2017	EU 28, CH	500 m	> 5 t soil $ha^{-1}a^{-1}$ (Panagos et al. 2015), limited to cropland
	Soil Organic Carbon (SOC) Saturation Capacity	Lugato et al. 2014a, 2014b	EU 28	250 m	<0.5 Ratio between actual and potential SOC stock (Lugato et al., 2014a, 2014b)
Water related Deficit Regions	Irrigation	Eurostat 2017; JRC Water Portal 2017	all Europe	100 m, 1000 m	>25% irrigated land
	Nitrogen surplus	BAFU 2015; Leip et al. 2014	EU 27, CH (without Cyprus)	1000 m (100 m CH)	>70 kg N ha^{-1} a^{-1} (BMUB, 2017).

Climate Risk Regions	Climate change / Temperature rise	Hijmans et al. 2005	all Europe		2 - 4° C between 1990 and 2050 (Hart et al., 2012)
Biodiversity Deficit Regions	Soil biodiversity	Combination of Soil Fauna, soil microorganisms and soil biological function; Orgiazzi et al. 2016	EU 27	500 m	Risk level "high", "moderate-high" (Orgiazzi et al., 2016)
	Landscape suitability to support pollination	Rega et al. 2017	all Europe (without Cyprus)	100 m	Classes "very low" and "low" (Rega et al., 2017)
	Pest control index	Rega et al. 2018	all Europe (without Cyprus	100 m	Classes "very low" and "low" (Rega et al., 2018), limited to cropland

5.2.2.5 Selection of Priority Areas

Using the thresholds previously mentioned (Table 8), the nine environmental deficits were spatially combined using GIS. In each spatial unit the number of deficits were added together. In the resulting "heatmap", the 10% of the area with the highest number of deficits was defined as the Priority Area for the implementation of agroforestry. Based on Mücher et al. (2010) the Priority Areas were clustered into seven biogeographical regions: Atlantic; Continental lowlands, and hills; Mediterranean lowlands, hills, and mountains; and Steppic.

The spatial analysis was performed in ArcGIS10.4 (ESRI 2016) and R (R Development Core Team 2013). The outcomes were performed in R (R Development Core Team 2013).

5.2.3 Agroforestry recommendations

Potential agroforestry practices, which were: 1) the most adapted to mitigate the prominent environmental issues in the region, 2) the most developed in the region and 3) the most suitable to face climate change, were compiled by local experts and the authors for each Priority Area.

With the aid of a structured template, the type of agroforestry (e.g. silvopastoral, silvoarable; hedgerows, coppice, or single trees), a short description of the system, tree and hedgerow species, planting scheme (e.g. lines, scattered) and management system (e.g. year of harvesting / harvesting cycles, tree products and associated crop combinations) were collated. The outcomes were summarized by biogeographical region.

5.2.4 Assessment of carbon sequestration in biomass

The total biomass production (aboveground wood and root biomass) of the woody elements and the carbon storage potential of the proposed agroforestry systems were assessed based on

literature data (see Annex III) and expert knowledge [units: t biomass ha^{-1} a^{-1}; t C ha^{-1} a^{-1}]. Herein the values represented an average potential per year of tree life and did not consider any dynamics of tree growth over time, or other impact factors such as water and nutrient availability, temperature, tree density, etc. Potential minimum and maximum values of carbon storage in biomass (both above and belowground) of each agroforestry practice for each biogeographic region were extracted separately for grassland and cropland. These values were used for upscaling the results to the "Priority Area", assuming that in those regions, the total available farmland would be converted into agroforestry with one of the recommended agroforestry practices.

5.3 Results

5.3.1 Deficit assessment

In EU27 and Switzerland, the total area of European agricultural land is 1,544,022 km^2. Subtracting existing agroforestry and nature protection areas, the analysis was then concentrated on 1,414,803 km^2 as Focus Area. This area consisted of 343,624 km^2 of pasture (88% of total European pasture) and 1,071,179 km^2 of cropland (≙ 92% of total European cropland).

Figure 21 gives an overview of the size of the individual "Deficit Areas" in relation to the Focus Area. Soil loss risks over 5 t soil ha^{-1} a^{-1} from water erosion were identified on 9.5% of the grassland and 11.9% of the cropland area. Areas suffering from an annual loss greater than 5 t soil ha^{-1} a^{-1} by wind erosion were relatively small (1.5%), whereas a low SOC saturation capacity was present on 12.8% of grasslands and 58.7% of croplands. In total, 1% of the grassland and 8.4% of the cropland had irrigation levels greater than 25%. High nitrogen pollution risk was mapped on 34.5% of the grasslands and on 20.6% of croplands. Around 53.6% of grasslands and 63.0% of croplands were located in regions where temperature is expected to rise between 2 and 4°C by 2050 according to HadGEM2-ES forecast scenario. Deficits in biodiversity and resulting potential underprovision of ecosystem services are widely spread all over European agricultural land. In total, 66.4% of croplands in the Focus Area were predicted to have low or very low natural pest control potential, whilst 21.0% of grasslands and 41.8 % of croplands were predicted to be not suitable for supporting pollinators. Potential soil biodiversity deficits were mapped on only 18.7% of grasslands and 11.5% of croplands.

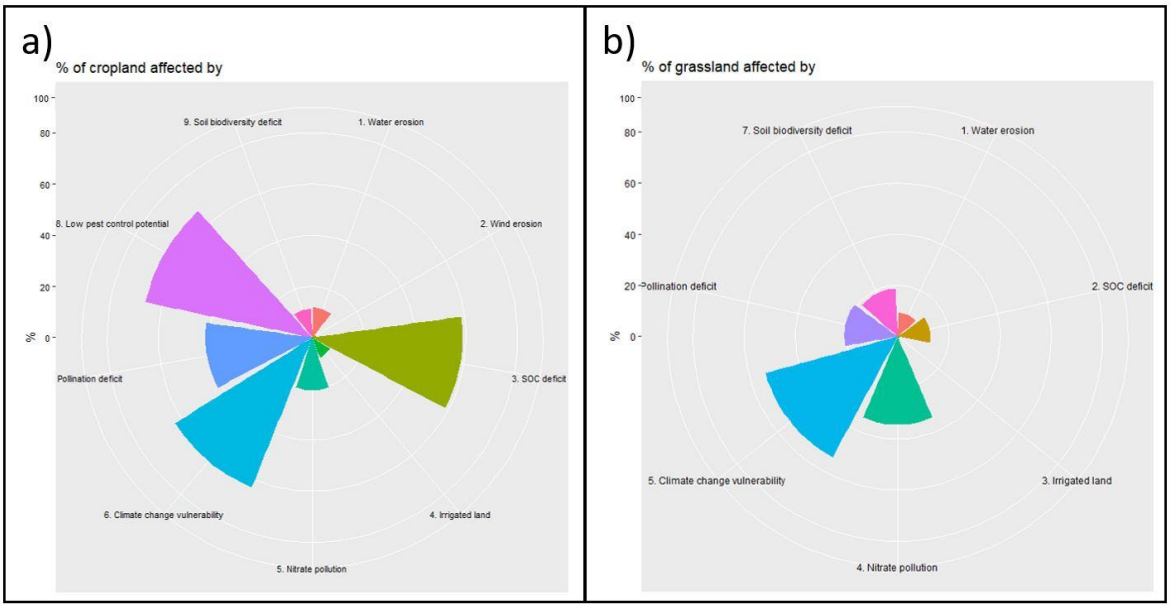

Figure 21: The proportion of a) the cropland and b) the grassland affected by each of the deficit pressures across the selected Focus Areas. Pest control and wind erosion were only considered in cropland areas. SOC: Soil organic carbon

By combining the nine individual deficit maps, we created a heatmap for environmental deficits (Figure 22 a). This area was targeted to pressures that can be mitigated by agroforestry.

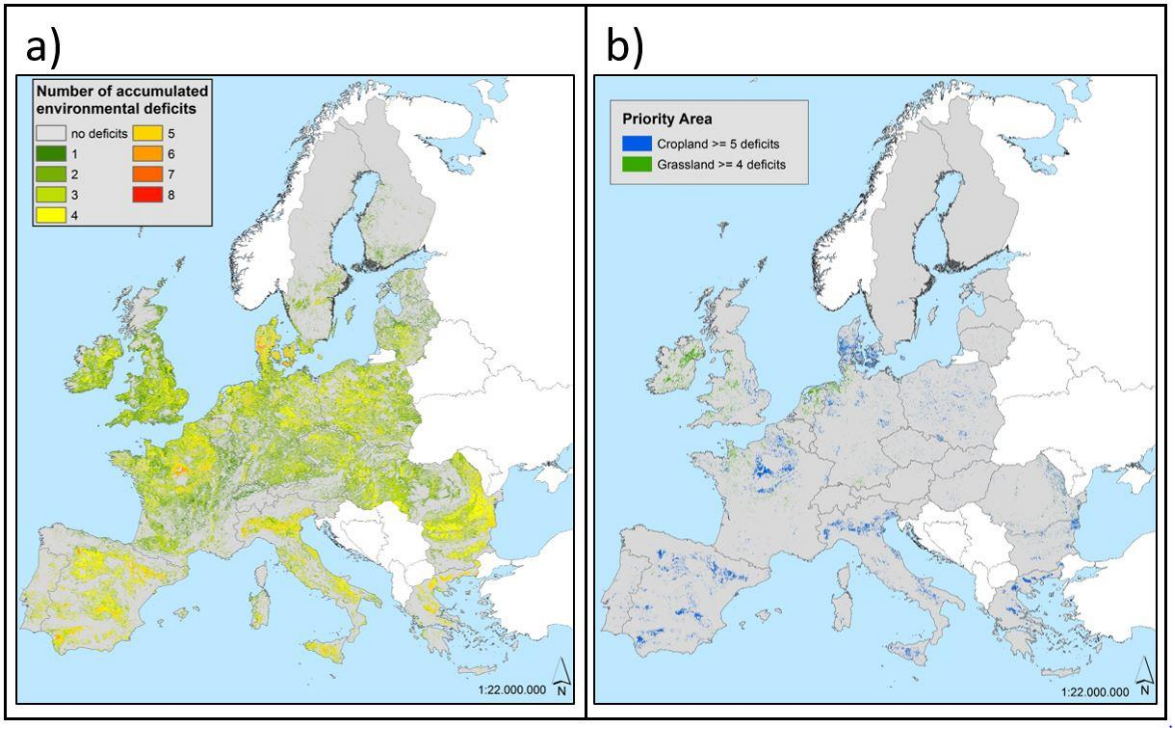

Figure 22: a) Heatmap for the number of environmental deficits and b) Priority Areas (Grassland areas with more than four deficit indicators (green) and cropland areas with more than five deficit indicators (dark blue)).

For the total Deficit Area, a lower proportion of grassland areas were identified than croplands. Only 4% of the croplands in the Focus Areas had no deficits, while in grassland it was around 12%. More than half of the grassland areas had less than three deficits, while 35% of cropland

were affected by more than four deficits, and 9% had more than five deficits. Whilst we defined the Priority Areas as croplands with more than five deficits, we set the threshold to only four deficits for grassland, as we evaluated only seven grassland deficit indicators (excluding soil erosion by wind and pest control deficit). Together, they represent the worst 10% of the Deficit Area (Figure 3b). These combined Priority Areas for cropland and grassland amounted to 136,758 km^2, which corresponds to about 8.9% of the total European agricultural land. Table 9 gives an overview of the Priority Areas according to country and biogeographical region.

Table 9: Summary of the Priority Areas by country divided into biogeographical regions based on the Landscape classification by (Mücher et al., 2010).

Biogeographical Region		**Country**	**Cropland [km2]**	**Grassland [km2]**	**Total [km2]**	**Share of total agricultural land [%]**
Atlantic		**Total**	**29,611**	**29,088**	**58,698**	**9.74**
		Denmark	498	3,223	3,721	20.19
		France	16,156	6,151	22,308	10.7
		Germany	6,366	102	6,468	9.78
		Ireland	6	7,133	7,139	17.12
		Netherlands	2,624	3,030	5,654	32.96
		UK	2,600	8,719	11,319	8.43
		others	1,361	730	2,090	1.8
Continental	**Lowlands**	**Total**	**7,644**	**1,259**	**8,903**	**6.24**
		Denmark	3,607	21	3,628	38.82
		Germany	1,660	809	2,469	5.22
		Poland	1,296	106	1,402	3.76
		Others	1,081	322	1,403	2.88
	Hills	**Total**	**13,906**	**4,360**	**18,265**	**4.11**
		Bulgaria	2,116	537	2,654	7.03
		Germany	1,905	1,473	3,377	3.88
		Poland	6,379	439	6,818	5.73
		Romania	2,054	1,078	3,132	4.87
		others	1,452	833	2,285	1.68
Mediterranean	**Lowlands**	**Total**	**12,399**	**156**	**12,555**	**22.52**
		Greece	3,020	42	3,063	38.28
		Italy	7,990	39	8,029	21.15
		Spain	1,220	50	1,270	22.36
		Others	169	25	193	4.7
	Hills	**Total**	**20,226**	**650**	**20,876**	**15.53**
		Greece	2,340	117	2,457	22.04
		Italy	6,985	83	7,069	15.64
		Spain	9,676	227	9,903	25.02
		Others	1,225	223	1,448	3.77

	Mountains	**Total**	**12,858**	**628**	**13,486**	**10.96**
		Italy	1,071	78	1,149	10.66
		Spain	11,176	429	11,606	12.34
		Others	611	120	732	4.02
Steppic		**Total**	**2,948**	**1,026**	**3,974**	**11.54**
Total			**99,592**	**37,166**	**136,758**	**8.87**

5.3.2 Potential agroforestry practices

In total, 64 agroforestry practices were proposed by the authors and local experts. They cover a wide range of practices from hedgerow systems on field boundaries to fast growing coppices or scattered single tree systems. Table 10 lists, for each biogeographical region, the proposed system with the lowest, medium, and highest carbon sequestration potential (see Annex III for the complete list). In line with the largest Deficit Areas, the highest number of agroforestry practices was proposed for Atlantic regions (14 silvopastoral and 9 silvoarable practices) followed by Mediterranean arable land.

Table 10: Agroforestry practices for cropland and grasslands in the European biogeographical region (Only an extract of the practices with the lowest, with medium and with the highest carbon sequestration potential are shown. See Annex III for the complete list and references). SRC: Short rotation coppice.

Biogeographical region	**Agroforestry type**	**Title**	**Tree / Hedgerow Species**	**Trees [trees ha^{-1}], Hedgerow [m ha^{-1}] or Wood Cover [% ha^{-1}]**	**Planting system**	**Crop Species and Products**	**Tree Products**	**Year of tree harvesting**	**Carbon sequestration, [t C ha^{-1}a^{-1}]**
Atlantic grassland	Silvopastoral, coppice	Agrofoestry for ruminants in France	Pear (*Pyrus* spp), honey locust (*Gleditsia triacanthos*), service tree (*Sorbus domestica*), white mulberry (*Morus alba*), Italian alder (*Alnus cordata*), goat willow (*Salix caprea*), field elm (*Ulmus minor*), black locust (*Robinia pseudoacacia*), grey alder (*Alnus incana*)	(single -2m, double -6m, triple -10m), 4m for trees, 1.3m coppices x 20m, (11% woody cover)	Single, double, or triple lines	Grazing, hay, silage	Fodder-trees, woodchips	5 – 8	0.16 - 0.48
Atlantic grassland	Silvopastoral, single trees	Traditional orchard	Fruit trees (apple – *Malus domcestica*, pear - Pyrus spp, plum - *Prunus domestica*)	80 trees ha^{-1}	Lines	Grazing, hay, silage	Fruits (woodchips)	60	1.23
Atlantic grassland	Silvopastoral, single trees	High stem timber trees	Poplar (*Populus* spp)	400 trees ha^{-1}, After 15-20 years: 120-150 trees ha^{-1}	Lines	Grazing, hay, silage	Timber	First cut: 15-20 harvest:25-30	2.78-6.35
Atlantic arable	Silvoarable, hedgerows	Productive boundary hedgerow	Mixed hedgerow species: hawthorn (*Crataegus* spp), blackthorn (*Prunus spinosa*), field maple (*Acer campestre*), hazel (*Corylus avellane*)	0.03 % ha^{-1}	Boundary hedgerow	Crop rotation with cereals (wheat, barley, oats), potatoes, squash, organic fertility building ley	Woodchips	Every 15	0.1 - 0.45
Atlantic arable	Silvoarable, coppice	Alley cropping – Short Rotation Coppice (SRC)	Willow (*Salix viminalis*), hazel (*Corylus avellana*)	1000 - 1300 trees ha^{-1} (24% ha^{-1})	Twin rows with 10-15m wide crop alley	Cereals (wheat, barley, oats), potatoes, squash, organic fertility building ley	Woodchips	Every 2 for willow, every 5 for hazel	0.36-1.05
Atlantic arable	Silvoarable, single trees	High stem timber trees	Walnuts (*Juglans regia*), maples (*Acer* spp), wild cherry (*Prunus avium*), checker tree (*Sorbus torminalis*), service tree (*Sorbus domestica*), apple (*Malus domestica*), pear (*Pyrus* spp).	28-110 trees ha^{-1}, (26-50 m between rows)	Lines		Timber	60	Walnut: 0.32 - 2.75, cherry: 0.19 - 1.4
Continental grassland	Silvopastoral, single trees	Wooded grassland	Fruit trees: cherry (*Prunus avium*), walnut (*Juglans regia*), apple (*Malus domestica*), etc.	60 trees ha^{-1}	Lines	Grazing, hay, silage	Fruits	70-90	Cherry: 0.41-0.76, apple: 0.93-1.43, walnut: 0.86 -1.16
Continental grassland	Silvopastoral, coppice	Agroforestry for free-range pig production	Poplar (*Populus* spp), willow (*Salix* spp), various fruit trees	10-40 % ha^{-1} (2.5x3.5m)	SRC lines	Grazing, hay, silage	Woodchips, fodder-trees	5-8	Poplar: 0.44-1.41
Continental grassland	Silvopastoral, single trees	High nature and cultural value wood pastures and wooded grasslands	Sessile oak (*Quercus petraea*), beech (*Fagus sylvatica*), hornbeam (*Carpinus betulus*), wild fruit trees, mixed poplar (*Populus* spp.), willow (*Salix* spp.)	50-300 trees ha^{-1} (10-50% ha^{-1})	Scattered	Grazing, hay, silage	Acorns, fruits, timber, (fodder-trees)	Trees not harvested	Oak: 0.71 - 2.83, beech: 0.59- 2.34, hornbeam: 0.38 - 1.55

Biogeographical region	Agroforestry type	Title	Tree / Hedgerow Species	Trees [trees ha^{-1}], Hedgerow [m ha^{-1}] or Wood Cover [% ha^{-1}]	Planting system	Crop Species and Products	Tree Products	Year of tree harvesting	Carbon sequestration, [t C $ha^{-1}a^{-1}$]
Continental arable	Silvoarable, coppice	Alley cropping	Poplar (*Populus* spp); Mixed hedgerow species: willow (*Salix* spp), hornbeam (*Carpinus betulus*), common ash (*Fraxinus excelsior*), common birch (*Betula pendula*), black locust (*Robinia pseudoacacia*)	Rows A, B, and C: 10'000 trees ha^{-1}, Rows D, E, F, and G: 2222 trees ha^{-1}, (10% ha^{-1}).	Single and twin rows with 48, 96, and 144 m wide crop alleys.	Crop rotation (wheat, maize, oilseed rape, barley)	Woodchips	Rows A, B, and C: every 3-5. Rows D, E, F, and G: every 8 – 10	0.15 - 0.44
Continental arable	Silvoarable, single trees	Orchard with vegetables or fruits (strawberries)	Fruit trees: cherry (*Prunus avium*), walnut (*Juglans regia*), apple (*Malus domestica)*, etc	60 trees ha^{-1}	Lines	Vegetable, berries (strawberries)	Fruits, timber	70-90	Cherry: 0.41-0.76, apple: 0.93-1.43, walnut: 0.86 -1.16
Continental arable	Silvoarable, single trees	Non-native, energy tree with Alfalfa	Pauwlonia (*Paulownia tomentosa*)	126 trees ha^{-1} (18 m x 5 m)	Lines	Triticale, alfalfa	Timber	10-12	3.77
Mediterranean grassland	Silvopastoral, single trees	Dehesa	Holm oak (*Quercus ilex*)	25-50 trees ha^{-1}	Scattered	Grazing	Acorns, fodder-trees	Trees not harvested	0.09 – 0.16
Mediterranean grassland	Silvopastoral, single trees	Grazed cork oak plantation	Cork oak (*Quercus suber*)	113 trees ha^{-1}, after 20 years: 50 trees ha^{-1}	Lines	Grazing	Cork, timber	80	0.34-1.29
Mediterranean grassland	Silvopastoral, single trees	Grazed fruit plantations	Olive (*Olea europaea*), almond (*Prunus dulcis*)	250 trees ha^{-1}	Lines	Grazing, legume rich mix (annual self seeding species)	Fruits, oil, nuts	Annual prunings, trees not harvested	Olive: 1.97, almond:1.36
Mediterranean arable	Silvoarable, single trees	High stem timber trees	Pedunculate oak (*Quercus robur*)	57 trees ha^{-1}	Lines	Cereals	Timber	35	0.11 -0.26
Mediterranean arable	Silvoarable, single trees	Fruit tree alley	Olive (*Olea europaea*)	200-400 trees ha^{-1}	Lines or scattered	Wild asparagus	Oil, forage	Annual prunings, trees not harvested	1.57-3.14
Mediterranean arable	Silvoarable, single trees	High stem timber trees	Poplar (*Populus* spp)	200 trees ha^{-1}	Lines	Crop rotation wheat, oilseed rape, chickpeas	Timber	15	5.76 - 7.29
Steppic arable	Silvoarable, single trees	High stem forest trees	Poplar (*Populus* spp), willow (*Salix* spp.), black locust (*Robinia pseudoacacia*), pedunculate oak (*Quercus robur*), plain common and black walnut (*Juglans nigra*), common ash (*Fraxinus excelsior*), red oak (*Quercur subra*),), lime (*Tilia* sp.),	60 – 70 trees ha^{-1}	Lines	Vegetables	Timber	70-90	Poplar: 1.72 - 2.85, oak: 0.32-1.2, walnut: 1.31
Steppic, arable	Silvoarable, single trees	Mixed timber and wild fruit species plantation	Grayish oak (*Quercus pedunculiflora*), field maple *(Acer campestre*), lime (*Tilia* sp.), hawthorn (*Crataegus* sp), *Rosa* sp, blackthorn (*Prunus spinosa*)	100 trees ha^{-1}	Lines	Vegetables	Fruits, fodder trees, timber	Harvesting depends on species estimated from 25 - 120.	Oak: 1.59, tilia:1.32

Biogeographical region	Agroforestry type	Title	Tree / Hedgerow Species	Trees [trees ha^{-1}], Hedgerow [m ha^{-1}] or Wood Cover [% ha^{-1}]	Planting system	Crop Species and Products	Tree Products	Year of tree harvesting	Carbon sequestration, [t C $ha^{-1}a^{-1}$]
Steppic, arable	Silvoarable, single trees	Poplar plantation	Poplar (*Populus* spp)	100 trees ha^{-1}	Lines	Sunflower, cabbage, corn, pepper and eggplant, water-melon and squash, cauliflower; wheat, beans	Timber	35	2.88 - 4.76

Table 11: Potential carbon sequestration in the whole Priority Area using minimum and maximum carbon storage potential of agroforestry practices proposed for each biogeographical region

Biogeographical region	Minimum carbon storage potential					Maximum carbon storage potential				
	[t C km^{-2} a^{-1}]		Priority Area [t C a^{-1}]			[t C km^{-2} a^{-1}]		Priority Area [t C a^{-1}]		
	Cropland	Grassland	Cropland	Grassland	Total	Cropland	Grassland	Cropland	Grassland	Total
Atlantic	10	16	296,109	465,401	761,510	275	635	8,142,998	18,470,618	26,613,616
Continental lowlands	15	44	114,660	55,396	170,056	159	141	1,215,401	177,518	1,392,919
Continental hills	27	38	375,461	165,661	541,122	377	283	5,242,545	1,233,741	6,476,286
Mediterranean lowlands	11	9	136,390	1,400	137,790	600	197	7,439,447	30,654	7,470,101
Mediterranean hills	11	9	222,488	5,850	228,338	530	197	10,719,872	128,053	10,847,925
Mediterranean mountains	11	9	141,441	5,650	147,092	729	197	9,373,711	123,676	9,497,387
Steppic hills	32	38	94,322	39,003	133,325	476	283	1,403,039	290,467	1,693,506
Total			**1,380,871**	**738,362**	**2,119,233**			**43,537,013**	**20,454,727**	**63,991,740**

5.3.3 Carbon storage potential

For each system the annual carbon storage potential of the woody elements (including roots) was identified using data from the literature and in each geographical region, the minimum and maximum storage potential were determined. The wide range of practices selected was echoed by a wide range of carbon storage potentials of between 0.09 to 7.29 t C ha^{-1} a^{-1}. In Table 11 these data were upscaled to the entire Priority Area of each biogeographical region.

Overall, implementing the proposed agroforestry systems in the Priority Areas could mitigate between 2.1 and 63.9 million t C a^{-1} depending on the systems chosen, which is between 7.7 and 234.8 million t CO_{2eq} a^{-1}.

In 2015, the 28 members of the European Union (EU28) together with Switzerland emitted 4,504.9 million tonnes of greenhouse gases (million t CO_{2eq}), with agriculture contributing 12% (~ 540 million t CO_{2eq}; Eurostat, 2017b). Converting the conventionally used farmland in the Priority Area (which was about 8.9% of total agricultural land) to agroforestry could therefore capture between 1.4 and 43.4 % of the European agricultural GHG emissions.

5.4 Discussion

This research investigated three questions: I) Where and to what extent is European agricultural land affected by (multiple) environmental pressures? II) Which regional types of agroforestry can be used to reduce environmental deficits? and III) What is the potential contribution of the proposed systems to the European zero-emission agriculture climate targets?

5.4.1 European environmental Deficit Areas

In response to the first question, several environmental pressures that can be mitigated by establishing agroforestry practices were selected. According to Alam et al. (2014) and Torralba et al. (2016) these include soil conservation, the improvement of water quality, nutrient retention, climate regulation, and enhanced biodiversity. We investigated nine environmental deficits and mapped their occurrence in European agricultural land, based on existing spatially explicit databases at a continental European scale. The best available data were used, although it should be noted that differences in scales (100 – 1000 m pixel size), time periods (2006 - 2017) and models (e.g. modelled soil losses in EU vs. soil erosion risk map in Switzerland) existed that might result in spatial inaccuracies (Schulp et al., 2014). All the datasets used, required some degree of modelling and the maps therefore showed a predicted rather than measured environmental deficits. Moreover, not all the existing environmental problems in

agricultural areas could be addressed. Methane emissions, ammonia emissions, and zoonoses contamination, for example, were not included in the analysis presented here. In addition, biodiversity aspects in terms of quality and diversity (Zhang et al., 2007), the amenity value of the landscape, and natural hazards, such as avalanches, floods, droughts, and landslides (EEA, 2017b) were not considered.

Recommendations from the literature were used to define the thresholds for delimiting the Deficit Areas. However, different thresholds exist and modifying these or using different models would affect the size and spatial location of the Deficit Areas. For erosion, we used 5 t soil ha^{-1} a^{-1} as a threshold for erosion caused by water and erosion caused by wind, whereas for example, adopting a "tolerable" soil erosion rate of 0.3 to 1.4 t soil ha^{-1} a^{-1} as recommended by Verheijen et al. (2009) would strongly have increased the Deficit Area. The 5 t soil ha^{-1} a^{-1} threshold was uniformly used for the whole of Europe. However, threshold soil erosion values could also be defined by the nature of the soils in a particular area, depending for example, on soil quality and depth, with lower quality and shallower soils given lower thresholds to reflect their already precarious state and the relative importance of conserving what remains.

Surplus regions for nitrogen have also been defined in different ways by the European states. Overall, the Nitrate Directive (91/676/CEE) limits the nitrate content in ground and drinking waters to 50 mg NO_3 l^{-1}, and uses this limit for national governments to identify Nitrate Vulnerable Zones (NVZ). In an earlier study on arable target regions for agroforestry implementation, based on soil erosion risk and NVZs, Reisner et al. (2007) identified 51.6% of the European arable land as Deficit Area. Yet the delimitation of NVZs was partly also a political process. In some countries they are limited to areas where the nitrate content in groundwater regularly exceeded the 50 mg NO_3 l^{-1} threshold. In other countries, entire territories or districts were designated where special actions for nitrate reduction are compulsory for farmers (European Commission, 2013b). For example, almost the entire territory of Germany is labelled as NVZ. To allow for a spatially more differentiated analysis, we opted to locate areas with modelled annual nitrogen surplus above 70 kg N ha^{-1} as a threshold. Together, they accounted for 22% of cropland and 36% of grasslands which is substantially lower than the 51.6% of European arable land identified by Reisner et al. (2007) as Deficit Area for nitrate emissions.

The most prominent deficit in terms of area affected was the impact of rising temperature and climate change. This is in line with Olesen et al. (2012) and Schauberger et al. (2017) who

modelled effects of climate change on crop development and yields. They found an earlier start to the growing and flowering period followed by enhanced transpiration in combination with water stress resulted in a reduction of maize yield of up to 6 % for each day with temperatures over 30°C. In fact, already during the summer of 2017 the potential impact of climate change was revealed by drought and heat waves, which impeded cereal production in various parts of Europe, mainly in southern and central Europe (JRC, 2017). However, by contrast, Knox et al. (2016) predicted positive effects of between 14-18% on the yields of wheat, maize, sugar beet, and potato by 2050 in Northern Europe.

To identify Priority Areas, we accumulated all indicators. This simple addition gave the same weight to all the environmental problems addressed. But, soil erosion could be more damaging for agricultural practices than pests in a particular region or vice versa. However, our methods and results are comparable with e.g. Mouchet et al. (2017) and Maes et al. (2015). Both analysed the ecosystem service provision of European landscapes. Mouchet et al. (2017) aggregated bundles of ecosystem services and found a longitudinal gradient of decreasing land use intensity from France to Romania. Maes et al. (2015) assessed the quantity of green infrastructure that maintained regulating ecosystem services and showed that regions with intensive agricultural production (arable and livestock) generally had lower levels of regulating ecosystem services provision. Both studies referred to the sum of all assessed indicators. The similarity among the three studies for the spatial output gives confidence to the overall outcomes of this study.

5.4.2 Potential agroforestry practices and ecosystem service provision

To address the second research question, the collection of agroforestry practices, we hypothesized that agroforestry could mitigate the environmental deficits identified and that for each region, suitable practices could be proposed. Although agroforestry provides multiple ecosystem services (Torralba et al., 2016), there is a general lack of uptake by farmers (Rois-Díaz et al., 2018). Therefore, instead of trying to propagate the most suitable agroforestry for a particular deficit area and environmental deficit, we argue that highest impact could be achieved by proposing agroforestry practices which are local-adapted and attractive for farmers. This was how the experts selected the proposed practices. The suitable combination of tree and crop species is highly dependent on soil, water and climate conditions at specific locations. For this reason, we have provided only a list of example agroforestry practices. The composition,

implementation, and management of the agroforestry systems needs to be discussed with regional agroforestry experts and developed in partnership with the farmers themselves[1].

For soil conservation, silvoarable alley cropping systems have been evaluated in earlier studies. Palma et al. (2007) and Reisner et al. (2007) estimated that their introduction on eight million hectares of cropland subject to water induced erosion risks, would reduce soil erosion in those areas by 65%. Similar findings were provided by Ceballos and Schnabel (1998) and McIvor et al. (2014), who analysed how agroforestry can contribute to soil protection and preservation. Hedgerow systems lowered wind speed and consequently soil erosion by wind (Sánchez and McCollin, 2015). Regarding the reduction of nitrate leaching, Nair et al. (2007) and Jose (2009) showed that agroforestry reduced nutrient losses by 40 to 70%. The conversion of 12 million ha of European cropland in NVZ to agroforestry with high tree densities could reduce nitrogen leaching by up to 28% (J. H. N. Palma et al., 2007). Moreno et al. (2016), Birrer et al. (2007), Bailey et al. (2010) and Lecq et al. (2017) investigated the potential of agroforestry to provide multiple habitats for flora and fauna and enhance biodiversity. Flowering trees, such as orchards with fruit trees, were especially important in providing nesting and foraging habitats for pollinators (Sutter et al., 2017) and could enhance pest control (Simon et al., 2011). And as a general rule, it has been found that green infrastructure, such as agroforestry, enhances the overall provision of regulating ecosystem services (Kay et al., 2018b; Maes et al., 2015).

5.4.3 Carbon sequestration potential

Our third research question focussed on the most prominent deficit “climate change” in pursuit of a zero-emission scenario in European agriculture. To do this, we estimated the carbon storage potential of the proposed agroforestry systems in the above and below ground biomass of the woody elements. Whilst we are aware that agroforestry can also increase soil organic carbon (e.g. Feliciano et al., 2018; López-Díaz et al., 2017; Seitz et al., 2017; Upson and Burgess, 2013), soil carbon storage is difficult to quantify at the scale we operate at.

We found an overall average carbon sequestration potential of agroforestry of between 0.09 to 7.29 t C ha^{-1} a^{-1}. The lower values were related to systems involving fewer woody elements per area (e.g. hedgerows on field boundaries which typically make up less than 5% of the field). The high values were mainly related to systems with higher densities of fast growing tree species and good soil conditions which would also be associated with some reduction in food and feed production (see also Table 3). Previous studies (e.g. Palma et al. 2007; Reisner et al.

[1] See also European Agroforestry Federation (EURAF) - http://www.eurafagroforestry.eu/

2007) estimated a sequestration range of between 0.77 and 3 t C ha^{-1} a^{-1} for alley cropping, and Aertsens et al. (2013) proposed an average sequestration of 2.75 t C ha^{-1} a^{-1}. Our estimates ranged from 0.09 to 7.29 t C ha^{-1} a^{-1} for implementing different agroforestry systems across Europe. In comparison, European forest stands sequestered 167 million t C in 2015 on 160.93 million ha (i.e. 1.04 t C ha^{-1} a^{-1}) (FOREST EUROPE, 2015). This value is a continental average and also comprises in trees grown at latitudes and altitudes where growth is relatively slow.

5.4.4 Potential implementation and impact

The hotspots of environmental deficits were mainly located in intensively managed agricultural regions mostly correlated with a high level of production (Eurostat, 2018, 2017c). The implementation of agroforestry in these regions would have the greatest environmental benefits (Weissteiner et al., 2016). In spite of the rising awareness of the importance of improving the environment and the investment in supporting measures of the European and national Rural Development Programs of the EU Member States (Santiago-Freijanes et al., 2018), the impact on green infrastructure is mixed. For example in the UK, whilst the area of woodland is increasing; the area of hedgerows declined from 1998 to 2007 (Wood et al., 2018). Agroforestry, landscape features, agro-ecological systems, and green infrastructure are still in decline (Angelstam et al., 2017; EEA, 2018; Salomaa et al., 2017). This implies that the established incentives are insufficient or do not adequately address the problem and actors (e.g. Mosquera-Losada et al., 2016). In contrast, a promising trend can be observed in Switzerland, where since 1993 agroforestry trees and hedgerows in open landscapes are qualified as ecological focus areas. This measure and the related payments have allowed consolidation of the number of Swiss agroforestry systems (BLW, 2017; Herzog et al., 2018).

There might be a trade-off between the introduction of agroforestry on arable and grassland, food production and the challenge of food security over the coming decades with a rising human population (Ray et al., 2013). For example, for a poplar silvoarable system in the UK, García de Jalón et al. (2017) predicted that crop yields would be 42% of those in arable systems, and that timber yields would be 85% of those in a widely-spaced forest system, i.e., the crop production and hence the production of food for human nutrition would be reduced. In the case of silvopastural practices, Rivest et al. (2013) showed that trees did not compromise pasture yields, though the impact of future drought pressures on yield would strongly be related to the chosen species.

The potential reduction of agricultural yields after the introduction of trees is an argument that is often put forward by farmers, who see themselves foremost as producers of food and fodder. However, under Mediterranean conditions, authors have also described that crop production could be reinforced under silvoarable schemes compared to open fields if the recurrence of warm springs keeps increasing (Arenas-Corraliza et al., 2018). In addition, farmers are increasingly being asked to provide environmental goods and services beyond food production and policy makers and researchers are seeking for ways to sustainably intensify agricultural production, which necessitates increasing productivity whilst at the same time reducing environmental damage and maintaining the functioning of agro-ecosystems in the long-term (Tilman et al., 2011; Tilman and Clark, 2014). In many cases, this will require a shift towards more complex and knowledge intensive agro-ecological approaches (Garibaldi et al., 2017). Trees on farmland have been identified for a long time as key elements in the design of sustainable agricultural systems (Edwards et al., 1993) and can contribute to multiple ecosystem services beyond carbon sequestration in combination with other types of semi-natural vegetation (Smith et al., 2017).

Agroforestry implementation in the Priority Areas, which made up only 8.9% of total European farmland, would capture between 1.4 and 43.4% of European agricultural GHG emissions, depending on whether the focus is on increasing tree cover in hedgerows as field boundary or supporting within field silvoarable and silvopastoral systems. These values support the observation by Hart et al. (2017) and Aertsens et al. (2013) who championed agroforestry as the most promising tool for climate change mitigation and adaptation. Consequently, agroforestry can contribute significantly to the ambitious climate targets of the EU for a zero-emission agriculture.

5.5 Conclusion

We investigated the potential for implementing agroforestry in environmental Deficit Areas of agricultural land in Europe and its contribution to European climate and GHG emission reduction targets. We found around one quarter of European arable and grassland affected by none or only one of nine analysed environmental deficits and not primarily in need of restoration through introduction of agroforestry. Grasslands were less affected than croplands. For the Deficit Areas, we proposed a wide range of agroforestry practices, which could mitigate the environmental deficits. The collection confirms the huge potential of agroforestry (1) to be introduced and established in nearly every region in Europe and (2) to adapt to various contexts,

ideas and needs of farmers. The estimated potential carbon storage depends on the selected agroforestry practice. The evidence from this study, that agroforestry on around 8.9% of European agricultural land could potentially store between 1.4 up to 43.4 % of the total European agricultural GHG emissions, is encouraging and demonstrates that agroforestry could contribute strongly to prepare the ground for future zero-emission agriculture. Future analysis should regionalize the approach to individual countries making use of data of higher spatial and thematic resolution, and ultimately to the farm scale, accompanied by extension and advice.

In sum, agroforestry can play major role to reach national, European and global climate targets, while additionally fostering environmental policy and promoting sustainable agriculture. Future policy and legislation, e.g. the future Common Agricultural Policy (CAP2020+), should explicitly promote and strenghen agroforestry.

Chapter 6

Synthesis

6.1 Conclusion

Agriculture, as one of the main land uses and key drivers of landscape changes (Plieninger et al., 2016; van der Zanden et al., 2016), faces multiple challenges nowadays and in the near future. The rising demand for high quality food and material should be satisfied in a sustainable and environmentally friendly way, while simultaneously adapting to changing climate and mitigating emissions and pollutions (Tilman et al., 2002). Moreover, the performance of agricultural land should not only be evaluated in relationship to its production function but also in terms of demands for environmental, regulatory, and aesthetic benefits from landscapes (Dale and Polasky, 2007). Sustainable and efficient agricultural production systems, also called "sustainable intensification", are needed (FAO, 2011; Petersen and Snapp, 2015; Tilman, 1999).

Agroforestry systems, the combination of woody elements on cultivated cropland or grassland (Somarriba, 1992), is one opportunity to address many of these targets (Hart et al., 2017; Jose, 2009). They provide food, fodder and timber, while simultaneously enhancing biodiversity and regulating ecosystem services at the plot level (Torralba et al., 2016). Moreover, they contribute significantly to the global carbon pool (Zomer et al., 2016) and constitute the land use systems with the greatest potential for climate mitigation and adaptation in European agriculture (Hart et al., 2017). Consequently, they could play a major role in future agricultural and climate policy to mitigate critical emissions. However, they are only located on 8.8% of European agricultural land (den Herder et al., 2017), and their existence in Europe is declining, mostly because of more profitable agricultural practices (Eichhorn et al., 2006; Nerlich et al., 2013).

Against this background, evaluating the economic and environmental impact of agroforestry practices at the landscape scale is the heart of the present thesis. The next sections puts forth the main findings in line with the underlying hypotheses (Chapter 1.5).

6.1.1 Methodological approach

This thesis analysed the ecosystem services supply of agroforestry systems from a landscape perspective by developing a spatially-explicit model (Figure 23). In undertaking this evaluation, indicators that characterise the ES delivery of agroforestry and agricultural systems were assessed. Evaluation focused on six ES indicators, namely biomass production and groundwater recharge rate as provisioning ES and the regulating services nutrient retention, carbon storage, soil preservation, and habitat and gene pool protection. The selection followed the Common

International Classification of Ecosystem Services (CICES) classification (Haines-Young and Potschin, 2013) with a focus on relevant indicators in agriculture and agroforestry systems.

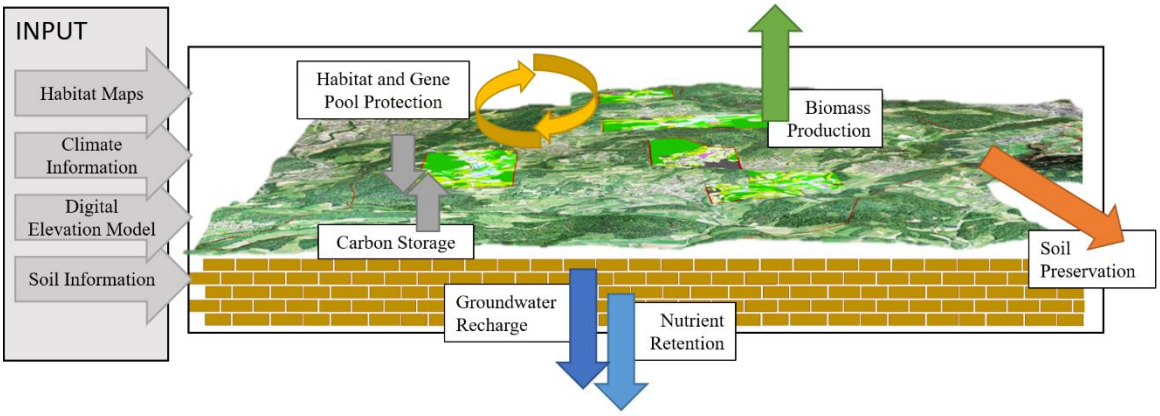

Figure 23: Conceptual model for the evaluation of the ecosystem services at the landscape scale

The key problem was to balance the methodology between model complexity, data requirements and total error (see e.g. Schröter et al. 2014; Palma et al. 2007). A reasonable level of detail to address the spatial effects of tree and crop interaction in agroforestry (e.g. Tsonkova et al. 2014; Prada et al. 2016) contrasted with impacts at the landscape scale (Maes et al., 2012; Mouchet et al., 2017). Landscape test sites (LTS) in contrasting landscapes (dominated by agroforestry versus dominated by agriculture) of 1 x 1 km spatial resolution were chosen. On this scale, both aims could be evaluated and visualised (Bailey et al., 2007a; Herzog et al., 2017).

Whilst the uncertainty of individual models could not be tested, the replication of their implementation across up to 96 landscape test sites made it possible to test the variability of the resulting indicators.

Based on the idea that computer simulation models can help to evaluate the long-term effects of different land use systems (Jose and Pallardy, 2004), existing algorithms for quantifying the indicators were identified, tested, adapted, and combined (see Chapter 2). The indicators, methods and data sources are: (1) For biomass production the EcoYield-SAFE model (Graves et al., 2010; Palma et al., n.d.) provided estimations of biomass of AF trees, crop yields and carbon sequestered. (2) The groundwater recharge was assessed using the water balance equation, which links precipitation, plant evapotranspiration, surface runoff and storage change in the soil. (3) The assessment of nitrate leaching was based on the water cycle modelling and was achieved by deploying the MODIFFUS 3.0 method. (4) The RUSLE equation (Renard et al., 1997) was applied to assess soil losses by water. (5) Carbon sequestration was estimated as

the sum of above and below ground crop and tree biomass, based on EcoYield-SAFE as well as the soil organic carbon (SOC), modelled in YASSO0.7 (Liski et al., 2005). (6) The habitat and gene pool assessment was divided into functions and capacities of nature represented by pollination and habitat richness and diversity. The Lonsdorf et al. (2009) equations were spatially applied in order to evaluate pollination potential for cavity and ground nesting species. As a pre-requisite, flowering and nesting facilities for wild pollinators were recorded during the habitat mapping. Landscape metrics, computed from LTS habitat maps, were used as proxies for habitat richness (Billeter et al., 2008), particularly the Simpson diversity index (SIDI), the share of semi-natural habitat (SoSNH) and the total number of semi-natural habitat types (ToSNH).

Indicators were evaluated based on the LTS habitat maps in combination with climate, soil and topographical information. The approach is limited by the availability and certainty of spatial data (Cushman and Huettmann, 2010; Lausch et al., 2015; Schulp and Alkemade, 2011) and by the state of the art of modelling, which reflects our current understanding of the relevant processes (Bailey et al., 2010; Kienast et al., 2009; Rykiel, 1996).

Finally, the algorithms were used to compare ES provision from agroforestry (AF) and non-agroforestry (NAF) LTS, using a traditional agroforestry system, namely cherry orchards in Switzerland, as an example. The resulting indicator values were largely plausible and within the range of values published in former (plot scale) studies.

This quantitative approach, a combination of field investigations and modelling, quantified provisioning and regulating ES at the landscape scale. The modelling approach is capable of capturing differences at the landscape scale.

6.1.2 Main findings

6.1.2.1 Differences between agroforestry and agricultural practices at the landscapes scale

The first research question inquired as to whether the provision of ecosystem services differed in landscapes with agroforestry compared to landscapes dominated by agriculture? The question was answered in conjunction with hypotheses HP1 and HP2:

(HP1): Agroforestry systems provide multiple ES and have an overall positive effect on conventional agricultural farming at the plot level (Alam et al., 2014; Torralba et al., 2016). Hypothesising that this positive effect of agroforestry radiates at the landscape level results in an overall higher provision of provisioning and regulating ES from

landscapes with agroforestry systems compared to landscapes with conventional agriculture.

(HP2) The beneficial impact of agroforestry at the landscape scale can be verified for various temperate agroforestry systems in Europe (Moreno et al., 2018; Pantera et al., 2018).

The spatial model described above was transferred to 12 European agroforestry landscapes (montado in Portugal, dehesa and soutus in Spain, groves in Greece, orchards in Switzerland, bocage in France, hedgerow landscapes in the UK and Germany, and wooded pastures in Romania, Switzerland and Sweden). Overall, the agroforestry systems in comparison to agricultural landscapes tended to deliver reduced nitrate losses, higher carbon sequestration, reduced soil losses, higher pollination services and higher porportions of semi-natural habitats. Higher annual biomass yields and higher groundwater recharge rates were linked to NAF areas. Figure 24 summarises the generell outcomes.

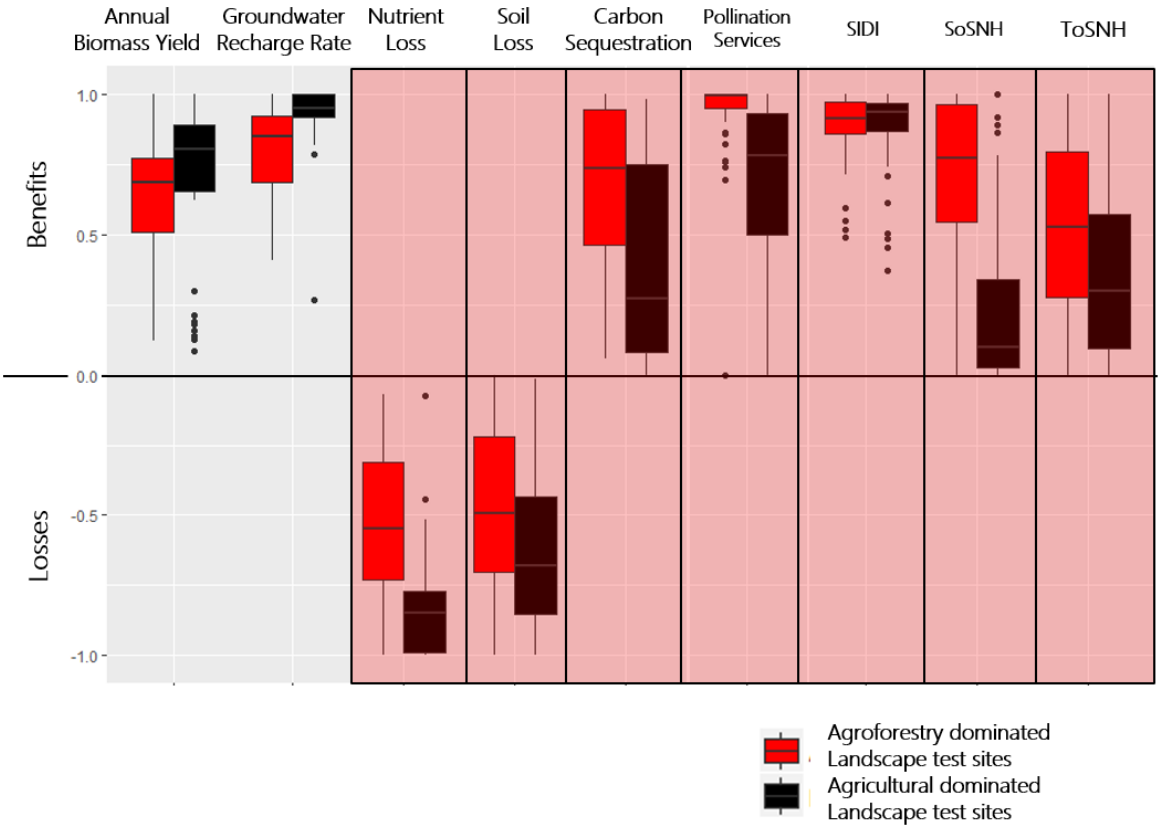

Figure 24: Summary of ES assessment grouped into agroforestry (AF - red) and non-agroforestry (NAF - black) landscape test sites aggregated over all case study region (n= 96 LTS). Indicators in red boxes perform better in AF LTS. Pollination services could not be evaluated for the UK. The bar graphs indicate mean values (horizontal line), standard deviation (upper and lower limits of boxes), range of values (lines) and outliers (points) [SIDI: Simpson's diversity index, SoSNH: share of semi-natural habitat, ToSNH: Total number of semi-natural habitats]

In traditional agroforestry landscapes, the provisioning ecosystem services were lower and less biomass was leaving the system per hectare and year (with the exception of Mediterranean agroforestry systems). A comparable reduced annual growth of AF systems is presented by Van Vooren et al. (2016). However, if the total lifetime of the systems (30 to 200 years) is accounted for, a higher total productivity of agroforestry in comparison to separate growing of trees and crops was shown by e.g. Sereke et al. (2015) and Graves et al. (2010). These findings may differ depending on the chosen agroforestry systems.

Regulating ES tended to perform better in AF landscapes. Significant differences were found for nutrient losses, carbon sequestration and share of semi-natural habitats. This was in line with the findings of, e.g, Nair et al. (2007) and Jose (2009), who showed that agroforestry systems can help reduce nutrient losses by 40 and 70%; moreover, Cardinael et al. (2015) and Zomer et al. (2016) found, somewhat similarly, that trees were contributing over 75% to the agricultural carbon pool. For the biodiversity metrics used here, differences were larger between case study regions than between AF and NAF LTS. This indicates the influence and relevance of broad landscape contexts in biodiversity assessments (Tscharntke et al., 2005). Studies by Birrer et al. (2007), Moreno et al. (2016b) and Bailey et al. (2010) support the conclusion that agroforestry landscapes are crucial for regional-specific biodiversity.

There was no significant difference between AF and NAF LTS for soil erosion. This disagreed with former studies, where AF systems were shown to prevent soil erosion (Palma et al. 2007; Rodríguez-Ortega et al. 2014; Sánchez and McCollin 2015). The AF LTS tended to have overall higher slope percentages. Standard multiple linear regression models were used to relate AF and NAF LTS (Figure 25), while p-values for slope were statistically significant and showed a reducing effect of AF on soil loss.

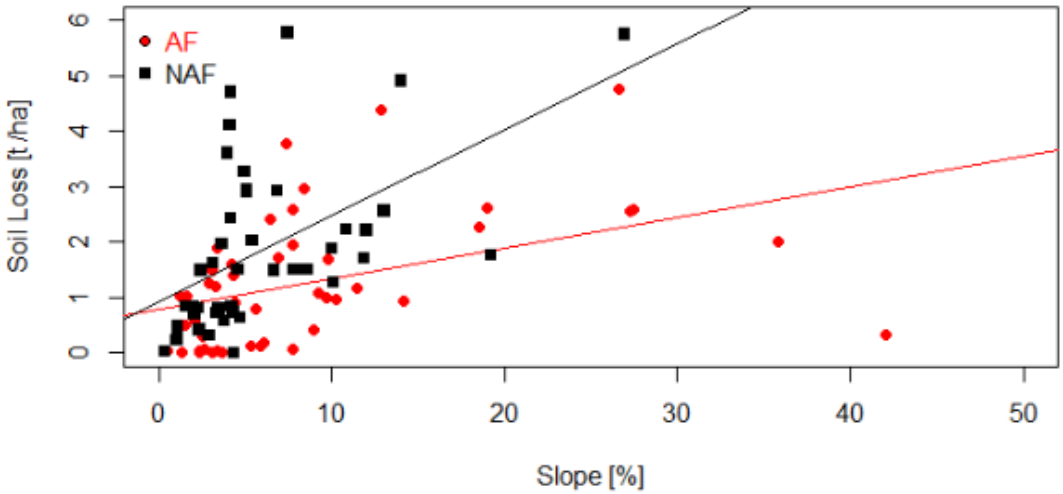

Figure 25: Erosion assessment grouped into agroforestry (AF, red) and non-agroforestry (NAF, black) for 96 landscape test (LTS) sites as a function of the slope. [p-value: 1.443e-06, Adjusted R2: 0.2625]

Overall, the analysis underlined that traditional agroforestry systems, regardless of type, region and composition had a beneficial impact on the provision of regulating ecosystem services at the landscape scale.

In summary, agroforestry landscapes enhance regulating ES provision. Particularly significant are the ES nitrate leaching, carbon sequestration and the share of semi-natural habitats. Provisioning ES, especially the annual biomass yield, are reduced in landscapes with agroforestry systems compared to agricultural dominated landscapes.

6.1.2.2 Evaluation of the economic and environmental impacts at the landscape scale

The second research question asked: is this ecosystem service provision related to economic and environmental benefits within these landscapes? This was interlinked with the following hypothesis:

HP 3: Valuing provisioning and regulating ES increase the profitability of landscapes with agroforestry and agro-ecological land management systems compared to agricultural landscapes (Alam et al., 2014; Zander et al., 2016).

One of the key elements of the ecosystem service framework, as presented in Chapter 1.3, is the valorisation and monetisation of ES. In Chapter 4 the economic and environmental benefits provided by landscapes (Chapter 1.2 and Chapter 1.3) were transferred into monetary units.

Currently, farmer income is mainly derived from agricultural production and European agricultural payments (European Commission, 2016). Additional environmental benefits from less intensive production methods, which conserve soil or retain nutrients, are not monetised, and are not valued (Ponisio et al., 2014). As Figure 26A illustrates, benefits from biomass production were reduced in Atlantic and Continental agroforestry landscapes. Mediterranean systems showed a much higher economic output. The agroforestry olive groves in our case study regions were already fully productive and therefore profitable. According to the European Commission (2012), olive production is one of the most important and profitable agricultural activities in southern marginal regions with poor productivity.

When valuing provisioning and regulating ecosystem services into a net landscape profitability, agroforestry systems emerged stronger in comparison to landscapes without agroforestry. There are additional profits from carbon capture and storage (Figure 26B), reduced pollution costs for nutrient emissions (Figure 26C) and soil losses (Figure 26D), together with a higher profitability compared to agricultural production. Already-low penalties per pollution unit (nutrient value > 2.5 EUR kg N-1; soil value > 17 EUR t soil-1) or additional payments per emission capture (carbon value > 30 EUR t C-1) would be sufficient to reach higher profitability than that being achieved by current agricultural production. These findings are echoed by Zander et al. (2016) in their assessment of the performance of grain legumes, and La Notte et al. (2017) in their assessment of in-stream nitrogen; indeed, said findings reflect the failure of markets to pass costs back to polluters.

Nutrient emission was the most important factor affecting the economic performance. Compared to this, soil losses were of lesser importance, even though the price per unit was higher (0.0 – 8.4 EUR kg^{-1} N versus 0.9 – 23.0 EUR t^{-1} soil). Similar results were obtained by García de Jalón et al. (2017).

The United Nations Global Compact (2016) proposes the use of a carbon value of \$100 t^{-1} (approximately 85 EURO t^{-1} C). The use of such high carbon values would dominate the economic performance of many land use systems. Even with a carbon price of 30 EUR t^{-1} C, landscapes with AF were more profitable compared to NAF LTS.

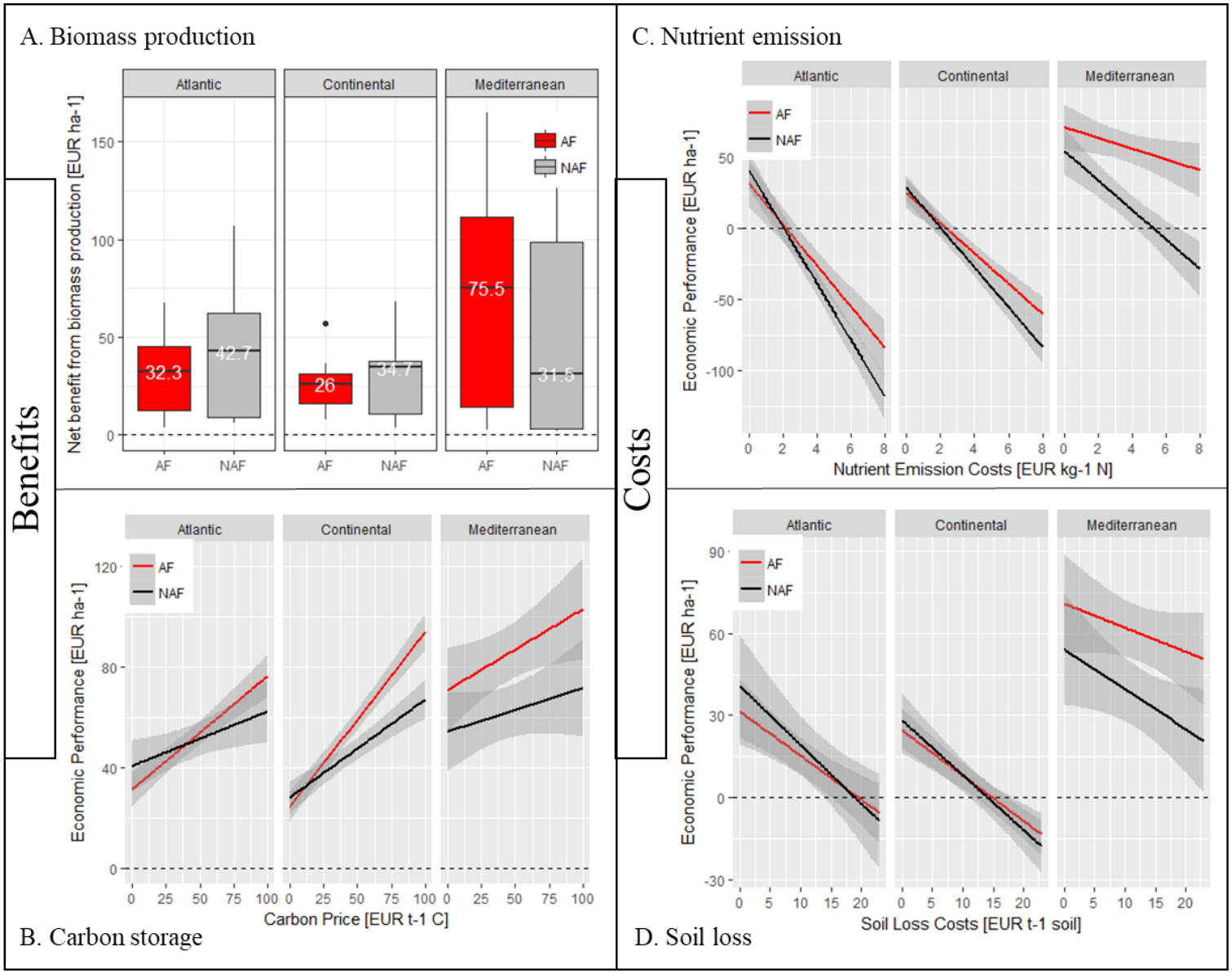

Figure 26: Economic performance of (A) biomass production benefits and for different (B) carbon prices, (C) nutrient emission cost and (D) soil loss costs together with the current sales revenues of biomass production in EUR ha^{-1} divided into biogeographical regions based on landscape test sites [LTS] grouped by land cover categories into agroforestry (AF) and non-agroforestry (NAF) sites

This analysis shows that the inclusion of non-market regulating ES can often result in AF landscapes, thus providing greater economic benefits to society than NAF landscapes across a range of biogeographic regions. This also demonstrates that there is a critical existing gap in economic assessments that fails to account for ecological benefits.

In average, the profitability of agroforestry landscape is reduced compared to agricultural landscapes. Valuing regulating ES increase the profitability of landscapes with agroforestry compared to agricultural landscapes.

6.1.2.3 Climate change potential of agroforestry at the continental scale

The third research questions tested the contribution of agroforestry systems to European climate targets of zero-emission agriculture. The related hypothesis was:

> HP 4: Agroforestry systems have a high climate change mitigation potential (in combination with other environmental and production benefits) in Europe (Alig et al., 2015; Hart et al., 2017).

As stated in the Introduction, one main motivation of this work was the great potential of agroforestry to contribute to climate change adaptation and mitigation. While the previous questions focussed on existing agroforestry and its impact on landscape, environment and society, this last question examined the potential effects of agroforestry planted on European agricultural land and its carbon storage potential.

Generally speaking, and as shown above, agroforestry systems provide several ecosystem services and mitigate environment deficits such as nitrate or soil losses (Kay et al., 2018b; J. H. N. Palma et al., 2007; Torralba et al., 2016). Against this background, environmental deficits were investigated spatially, based on nine deficit indicators (affected by wind or water soil erosion, nitrate surplus, irrigation, temperature rise up between 2 - 4°C, reduced soil organic carbon, soil biodiversity, pollination services and pest control). After accumulating the individual indicator maps into a “heatmap of environmental deficits”, the worst 10% were selected as a European Priority Area. Herein, agroforestry can be most beneficial (Reisner et al., 2007; Weissteiner et al., 2016). Once this had been accomplished, regional experts were asked to propose agroforestry systems suitable for the different Priority Areas. In total, 64 agroforestry systems were collected. The recommendations were related to tree and hedgerow species, suitable crops, management, and configuration of the system (e.g. number of trees or percentage of woody elements per hectare), and the potential carbon capture. The collection covered a wide range of systems, including hedgerow on field boundaries, coppice systems as

fodder-trees or for energy purposes and alley cropping systems with high stem trees. The minimum and maximum carbon storage potential were upscaled to the whole Priority Area.

The most remarkable findings were as follows: (i) the carbon storage potential per plot covered a wide range between 0.09 and 7.29 t C ha^{-1} a^{-1} and (2) the total carbon storage potential in the Priority Area could account for 1.4 and 43.4% of European agricultural GHG emissions. The lower values of agroforestry are related to AF involving less woody elements per area (e.g. hedgerows on field boundaries, which make up less than 5% of the field). The high values are mainly related to AF with fast growing trees species and good soil conditions on agricultural land. Previous studies (e.g. Palma et al. 2007; Reisner et al. 2007) explored a carbon storage range between 0.77 and 3 t C ha^{-1} a^{-1} and European forest stands sequester on average 1.04 t C ha^{-1} a^{-1} (FOREST EUROPE, 2015).

These outcomes are in agreement with Hart et al. (2017) and Aertsens et al. (2013), who stated that AF is the most promising tool to climate change mitigation and adaptation. Consequently, the findings demonstrate that agroforestry has the potential to mitigate and adapt to the challenges of climate change and might secure an unremitting and sustainable agricultural production in the future.

Finally, agroforestry has a high climate change mitigation potential. It contributes significantly to European climate targets of zero-emission agriculture.

6.1.3 Critical reflection of the theoretical approach

The theoretical backbones of the thesis are the landscape analysis and the ecosystem service concept. As described in the previous chapters, landscape analysis started with a focus on ecology (Turner, 1989), addressing the objective of enhancing biodiversity and environment by assuming a more diverse heterogenetic mosaic as proxy for greater biodiversity (Leopold, 1933). Disappointingly, the overall success was limited (Bailey et al., 2007a; Gonthier et al., 2014). The exclusive concentration on spatial structures overlooked other relevant elements and processes. Many attempts have been made to solve the problem, mainly by including additional elements and shifting the perspectives (Hein et al., 2006; Pardini et al., 2010).

Both factors, additional elements and changes of perspectives, were picked up by the ecosystems service concept in 2003, which addressed all indirect and direct contributions of ecosystems to sustainable human well-being (Costanza et al., 2017; MEA, 2003). In the following years, various approaches have been put forward to define, classify, map, assess and

value ES. It has even been placed on the political agenda and finally was incorporated in the European Biodiversity Strategy 2020 (European Commission, 2011). Silvertown (2015) and Boerema et al. (2017) pointed out two major drawbacks, asking, “Are ES oversold?” and “Are ES adequately quantified?” A lack of accuracy and consistency between studies and indicators are criticised. Moreover, besides the general problem of data uncertainty, there is still considerable ambiguity about how to link ecosystem characteristics to ecosystem services (Wong et al., 2015). This discussion is ongoing.

Finally, a challenging area in this field is the valuing of ES. The valuation process can mainly be divided into two stages, which are closely related. The first stage reflects the valuation unit. Several authors such as TEEB (2010), Sagebiel et al. (2016) and Zander et al. (2016) have assess monetary values for single ES. A more theoretical concept was presented by Costanza and Folke (1997). They propose a valuation along the three primary goals of efficiency, fairness and sustainability, wherein individual preferences, community preferences and the whole system preferences have to build a consensus. The second stage is about balancing between different types of ES. For example, is food and fodder production more important than purified water? Is there a higher need for flood protection than for recreation?

At this point, the landscape level comes into play again. Exceeding the thus-far descriptive landscape assessment, Sayer et al. (2013) and Minang et al. (2014) proposed a landscape approach based on a strong stakeholder involvement to reach multiple objectives by an adaptive management. The joint elaboration and the development of regional solutions are the key issues of this approach.

This leads to the scientific dilemma between landscape and ES research. On the one hand, the ES concept aims to harmonise ES indicators and assessment methods (e.g. Boerema et al. 2017; Englund et al. 2017), while on the other hand, the landscape approaches favour regional ES valuation schemes and priorities (Duguma et al., 2014; Scherr et al., 2012). Both research fields can present valid arguments for their concepts, and yet, current policies such as the FAO's climate-smart agriculture concept (FAO, 2017a) and the European Common Agricultural Policy (European Commission, 2016) tend to prefer stakeholder involvement and regional solutions.

Returning to this thesis, the approach used also struggles to fulfil all these presented requirements. The selected indicators, the underlying proxies, the methodologies, the models, and datasets were qualified as the best available data and state-of-the-art methods. However,

this approach is not exhaustive. Further indicators, a higher level of detail, and additional field data could have been beneficial. Moreover, stakeholder involvement was limited to some regional experts. Regarding the aim of this study to analyse twelve different agroforestry landscapes all over Europe using a comparable methodology, this method suffers from several pitfalls regarding the regional explanatory power. A broader regional participation would have benefited the outcomes.

Overall, ES is a "living" research field. Definitions, indicators, and methods are not yet settled. This provides high uncertainty and multiple opportunities.

Moreover, discrepancy exists between the targets of the ES concept to harmonise indicators and assessment methods and the aims of landscape approaches of joint elaboration and the development of regional solutions. There is no "one-size-fits-all" solution.

6.2 Outlook

With regard to the practical implementation of the findings, the next section provides ideas along the underlying questions: "How do we manage multifunctional landscapes?" (Sayer et al., 2013) and "How do we maintain landscape integrity?" (Plieninger et al., 2016, 2015b). Ideas arise from two viewpoints: (a) the top-down approach focussing on landscapes and the management level and (b) the bottom-up approach concentrating on agroforestry systems and the production level.

6.2.1 New ways for practical implementation: Landscape – management level

It is well documented that the horizontal and vertical structure of trees and woody elements outside forests enrich the environment and provide multiple ES services (e.g., Schnell et al. 2015; Torralba et al. 2016). This thesis proved that the ES benefits of trees in open landscapes - if they are valued - are not limited to their growing stand but have an overall beneficial impact on the surrounding landscape. They reduce pollution from nutrient and soil losses, mitigate climate changes and enhance the regulation of the biotic environment. These results have further strengthened the conviction that agroforestry – often overlooked as a niche of agricultural practice – provides multiple services and should be supported and extended. Integration into existing landscapes and their management is necessary.

Landscape management is best organised by regional stakeholders who are familiar with their demands and their local circumstances in combination with national stakeholders who provide

the broader context. One key part in this process is the involvement of as many stakeholders as possible through engagement, investment, and interventions. Relevant stakeholders include representatives from agriculture, tourism, education, health, small and medium enterprises, forests, landowners and municipalities. Examples demonstrated a strong and transparent stakeholder linkage combined with a polycentric governance during the planning and management phase facilitate the establishment and maintenance of new ideas (e.g. Scherr et al. 2012; Minang et al. 2014). A strong economic and social cohesion within the local society and a high attachment to the place, including identity and rights, will consolidate the process. The exchange of best practice examples – even across landscape and national borders – can drive and speed up the overall implementation.

In this context, the topic of land tenure is relevant, too. In fact, in 2013, only half of the agricultural land in Europe was owned by farmers and this share is decreasing (Eurostat, 2017d). Planting trees and preserving money for several years to land one does not own is challenging. Including all relevant stakeholders in the planning and management process will also mean including landowners. This can improve mutual understanding between farmers and landowners and might promote the idea of sustainable and long-term production with agroforestry and agro-ecological elements.

Against this background, a logical corollary of agroforestry practices are land sharing concepts. Two main ideas, spatial and temporal land sharing, can be distinguished.

Temporal land sharing has a long tradition in the context of animal grazing, also known as transhumance. In many parts of Europe, cattle, goats and sheep herds were seasonally moved from one (wooded) pasture to the next to satisfy their fodder demands (Mack et al., 2013; Olea and Mateo-Tomás, 2009). In the context of agroforestry innovative ideas to temporally combine different livestock practices could be beneficial. E.g. while free-ranged poultry systems value shade and shelter provided by trees during summer pig farms aim for additional fodder provision by nut or fruit trees in autumn. Ideally, the agroforestry system meets both these requirements.

While temporal (seasonal) land sharing is well known, spatial land sharing requires new conceptions. One idea could be to divide the business among the professions involved such as the agricultural and the forest profession. E.g. in Eastern Spain the separation of production tasks is very popular in part-time citrus farming (Picazo-Tadeo and Reig-Martínez, 2006) where management and entrepreneurial decisions such as fruit picking, fruit sales, pruning, or

ploughing are organised by traders or external workers. This externalisation of agricultural tasks is known as outsourcing and ranges from singular production tasks to whole business parts. Especially in smallholder farms the outsourcing of specialised mechanization for e.g. harvesting can be economically valuable (Zhang et al., 2017). The land sharing approach is partly similar, hence land ownership and / or the operational risk of the business would be shared too. The *usufruct* is a real right to derive profit e.g. the harvesting and using of the fruit production of agroforestry trees. Alternatively, there is the *co-ownership*, a concept in which two or more co-owners share the ownership of the land or property. The sharing would enable the sharing partners to stick to their main competences and responsibilities and benefit from the other partners and their competences. Ideally, this results in a win-win situation.

Taken together, the beneficial impact of agroforestry systems on pollution reduction and environment enhancement radiates to the landscape level. Regional stakeholders' interactions with various stakeholder groups (tourism, education, health, etc.) can support sustainable landscape management.

Land-sharing approaches can enhance landscape synergies. Innovative ideas for temporal and spatial land sharing established at the landscape level might provide additional benefits.

6.2.2 New ways for practical implementation: Agroforestry - Production level

Currently, farmers' income is mainly derived from agricultural production and European agricultural payments. Less intensive agricultural land management, with improved environment benefits, is often not as financially profitable under current subsidies and prices as intensive production (Ponisio et al., 2014). There is satisfactory agreement between these statements and the research outcomes of this thesis. 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 monetary values for the associated ES were included, the relative profitability of agroforestry increased. Similar findings for other agroecological practices are presented by e.g. Wezel et al. (2014) and Zander et al. (2016).

The Rural Development Programs of the European states took first steps by providing financial support for trees on agricultural land to farmers (Santiago-Freijanes et al., 2018). Despite this funding, landscape elements are still removed and the segregation of agricultural land into either

highly intensified or totally abandoned is ongoing (Biasi et al., 2016; Plieninger et al., 2016). This apparent lack of practical realisation and implementation of agroforestry systems demonstrates that either the amount of money is too low to cover the additional costs or farmers are not only concerned about money. This is in line with the findings of Sereke et al. (2015), who investigated the drivers and barriers to establishing agroforestry in Switzerland and found that farmers mainly fear for their social reputation. García de Jalón et al. (2017) and Rois-Díaz et al. (2017) added that increased labour, complexity of work, management costs and administrative burden were the biggest obstacles for agroforestry implementation. Taken together, new and innovative approaches to motivate farmers need to be evaluated. Particular attention should be paid to reducing administrative burdens and boosting the social recognition of agroforestry farmers.

As shown in Chapter 4, payments for ecosystems services improve the overall profitability of agroforestry systems. This is mainly due to the reduced pollution costs in agroforestry compared to agricultural production. Creating a marketable value per unit pollution reduction might encourage i) establishing a market with trade and sales, ii) raising awareness of environmental costs, and iii) finding the most cost-effective ways of reducing overall emissions and pollutions. The carbon market is one example. It is divided into two parts, emissions trading and the REDD+ (Reducing Emissions from Deforestation and Forest Degradation and the role of conservation, sustainable management of forests and enhancement of forest carbon stocks in developing countries) initiative. The European Union Emissions Trading System was launched in 2005 to reduce greenhouse gas emissions by penalizing emissions via certificates or "allowances". In contrast, since 2006, REDD+ has rewarded forest owners for capturing carbon and the forest stock. Both carbon valuation instruments are mainly market-based and therefore ideally financially self-supporting. Water pollution and soil degradation could be valued in a similar way if polluter and effects are spatially confined. In addition, alternative opportunities to finance agroforestry are crowdfunding, tree godparents or participative farming.

Finally, besides the fact that the profitability of agroforestry would increase, overall awareness and acceptance of these environmental friendly and climate-smart systems would raise and ideally be accompanied by enhanced implementation.

The global challenges of sustainable agriculture, which feeds the world and mitigates climate change, cannot be solved simply by introducing agroforestry. Hence, regarding

the idea "think global, act local", this thesis evaluated the effects of agroforestry on landscape scale and directed action to sustainable and climate-smart landscapes.

Lists of References, Figures, Tables

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and six manuscripts currently under review.

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Appendix

Annex I

Landscape modelling

Biomass production

Biomass production was calculated separately for (I) agricultural land, (II) forest areas and (III) agroforestry systems. The indicator was divided into the total stock value at any one time (t DM ha^{-1}), and the annual use (t DM ha^{-1} yr^{-1}) of the biomass (Table A1). No distinction was made regarding the type and quality of biomass. The biomass yields for arable crops were derived from Swiss agricultural statistics (BAFU 2013), grassland yield was from (AGRIDEA and BLW, 2017), and forest yields were from Swiss forest statistics (Brändli 2010, BAFU; BfS 2015a; BAFU; BfS 2015b). The EcoYield-SAFE model (Palma et al., n.d.) a daily time-step model developed from the YieldSAFE model (van der Werf et al., 2007), was calibrated to local monoculture yields of trees, crops, and grass, using the approach described by Graves et al. (2010), and then used to calculate the agroforestry biomass yield. The calibration data for EcoYield-SAFE were derived from field measurements, the literature listed above, local soil and weather data for a rotation of 60 years. The field data included tree and crown diameter, flowering time and fruit yield data. Information for grass yield, 6.0 t ha^{-1} for high-input grassland and 2.0 t ha^{-1} for low-input grassland, came from the literature (AGRIDEA and BLW, 2017) and from local farmers. The fruit yields from the cherry trees were assumed to be 50 kg (16 % DM) per tree for small trees, 100 kg per tree for medium sized trees, and 150 kg per tree for large trees in years with good weather conditions (Windisch, 1895). CliPick (Palma, 2017) provided daily data on precipitation, temperature and solar radiation. Soil parameters were taken from the description of hydraulic properties of European soils (Hiederer, 2013a; Wösten et al., 1999). For an agroforestry system of 80 cherry trees ha^{-1}, EcoYield-SAFE predicted 130 t of biomass in year 60, with a mean yield of 2.16 t ha^{-1} yr^{-1}. The mean fruit production was 16 t ha^{-1} yr^{-1} and grass production in between trees declined from approximately 6 t DM ha^{-1} in the first years to 2 t DM ha^{-1} in as the trees got older.

Table A1: Average biomass production and biophysical yield in tons dry matter per land cover class. Conversion: 40 t potatoes ha^{-1} (19 % DM) (BAFU 2013); 80 t sugar beet ha^{-1}(15 % DM)(BAFU 2013), 2 m^3 / t atro wood (Riegger, 2008).

Category	Crop	Stock (t DM ha^{-1})	Use (t DM ha-1 yr-1)	Source
Annual crops	Cereals		10	BAFU 2013
	Maize		17	
	Rape		3	

	Potatoes		7.6		
	Sugar beet		12		
Grassland	Grassland	2	5		(AGRIDEA and BLW, 2017)
Forest	Forest	175	3.25		Brändli 2010, BAFU; BfS 2015a; BAFU; BfS 2015b
Agroforestry	Trees		cherries	wood	Results EcoYield-SAFE
	- *young*	5	0	0.64	
	-*medium*	20	1	1.28	
	-*old*	30	2	2.56	
	Grassland	2	3		

Groundwater recharge rate

The general water equation was given as:

$$P = E + R + \Delta S \qquad \text{with } \Delta S = (\Delta S_{Soil} + \Delta S_{Groundwater\ recharge}) \text{ (Equations 8, 9)}$$

where P was the precipitation, E was the evapotranspiration, R was the surface runoff and ΔS was the storage change in soil (ΔS_{soil}) and groundwater ($\Delta S_{Groundwater\ recharge}$). Water flows were modelled by using FAO's CROPWAT 2.0 for crop performance indices (Allen et al., 1998) in combination with the spatial components of MODIFFUS 3.0 method (Hürdler et al. 2015). Our focus was on the amount of groundwater recharge and leachate as ES. The water cycle was calculated in six steps: (I) P as mean annual precipitation in mm was derived from the CCM River and Catchment Database compiled by the European Commission and Joint Research Centre for the years 1975 to 1999 (Vogt et al. 2007). The data were interpolated on a 1 km grid. (II) E was estimated by multiplying effective rainfall and reference evapotranspiration coming from the FAO CROPWAT 2.0 model as monthly values and leading to a discrete crop evapotranspiration (ET_C) using the Penman-Monteith (FAO-56 PM) method (Allen et al., 1998). (III) R was modelled using method of the MODIFFUS 3.0 (Hürdler et al. 2015), which incorporated slope as derived from the digital elevation model SwissALTI3D (swisstopo, 2012), land use characteristics and water catchment areas (Vogt et al. 2007). (IV) ΔS was divided into ΔS_{Soil} and $\Delta S_{Groundwater\ recharge}$. ΔS_{Soil} was obtained combining data on the total available water content (TAWC) for topsoil in the European Soil Database (ESDB) (Hiederer, 2013b, 2013a; Panagos et al., 2012), storing and filtering capacity in Makó et al. (2017) and the available water content (AWC) in Ballabio et al. (2016). (V) The soil water balance was calculated to provide the $\Delta S_{Groundwater\ recharge}$. (VI) The groundwater recharge rate (*GWRR*) represented the proportion of precipitation percolating to the groundwater (Equation 10) and was given as:

$$GWRR = \frac{\Delta S_{Groundwater\ recharge}}{P} * 100 \qquad \text{(Equation 10)}$$

Nutrient retention

The nutrient retention assessment was based on MODIFFUS 3.0, an empirical model for nitrate and phosphorus losses in Switzerland (Hürdler et al. 2015). In this model, leaching values for each land cover class were weighted by factors for soil characteristics, fertilizer application, grassland management, denitrifcation and drainage. The natural N background exposure was considered to be equal for all habitats and was therefore not incorporated into the calculation. N leaching was then multiplied by total leachate, and P loss was multiplied by runoff water, both of which were calculated in the water cycle assessment.

Soil preservation

The Revised Universal Soil Loss Equation (RUSLE) (Renard et al., 1997) (Equation 11) is defined as:

$$A = R * K * L * S * C * P \qquad \textit{(Equation 11)}$$

where A was the estimated mean annual soil loss (t ha^{-1} yr^{-1}), R was a rainfall-runoff erosivity factor, K was a soil erodibility factor, LS was a slope-length factor, C was a cover-management factor and P was a practice-management factor.

The *R* and *K* factors were derived from Panagos et al. (2014) and Panagos et al. (2016). The slope-length factor was calculated using the System for Automated Geoscientific Analyses (SAGA) (Conrad et al., 2015; Olaya, 2004) with digital elevation data from SwissALTI3D, which has a spatial resolution of 2 m. The *C* factor was defined for each habitat according to Panagos et al. (2015) and Hürdler et al. (2015). The *P* factor was set to 1, as no special supporting practice was used (Panagos et al. 2015).

Carbon storage

Our assessment of carbon storage is based on the produced above and below ground biomass estimated in EcoYield-SAFE. The carbon content of trees was assumed to be 50% of tree biomass (Aalde et al., 2006). Moreover, soil carbon storage was assessed by Yasso07 (Liski et al., 2005), which has been developed for evaluating tree and forest systems, agroforestry, grassland, and coppice systems (Masera et al., 2003; Prada et al., 2016). The Yasso07 model is able to address the decomposition of biomass fractions, and their effects on soil carbon, and

simulates the stock, annual change, and releases of carbon to the atmosphere based on site specific climate and stand information.

Habitat and gene pool protection

Pollination service was assessed based on an adapted Lonsdorf Model using ArcGIS. It consist of three equations; the habitat nesting suitability (HN_{sx}), the habitat flowering suitability (HF_{SX}) and the pollinator source (P_{OS}).

$$HN_{sx} = \sum_{j=1}^{J} N_{js} p_{jx} \qquad \textit{(Equation 12)}$$

where N_{jS} is the compatibility of land cover j for nesting bee species s in percent. The nesting capacity was divided into ground and cavity nesting facilities. Ground nesting facilities were assumed to exist in semi-natural habitats (SNH), medium intensive grassland and forest. Cavity nesting potential was assumend in all habitats with woody elements like hedergerows, agroforestry and forest.

$$HF_{sx} = \sum_{k=1}^{K} w_{sk} \frac{\sum_{m=1}^{M} \sum_{j=1}^{J} F_{js,k} p_{jm} e^{\frac{-D_{mx}}{a_s}}}{\sum_{m=1}^{M} e^{\frac{-D_{mx}}{a_s}}} \quad \text{with} \quad \sum_{k=1}^{K} w_{sk} = 1 \qquad \textit{(Equation 13)}$$

Herein p_{jm} stands for the proportion of parcels with land cover j, D_{mx} is the Euclidean distance between parcels m and x and a_s represents the foraging distance for species s. $F_{js,k}$ is the suitability for foraging of land cover j for species s during season k. We used the amount of clover and herbs in grasslands, crops pollinated by insects (mainly rapeseed, horticulture and vegetable production) and blossoming trees as flowering potential.

$$P_{os} = \frac{\sum_{m=1}^{M} P_{sm} e^{\frac{-D_{om}}{a_s}}}{\sum_{m=1}^{M} e^{\frac{-D_{om}}{a_s}}} \qquad \textit{(Equation 14)}$$

wherein P_{sm} is the relative abundance of pollinators on map unit m, D_{om} is the distance between map unit m and farm o and a_s is the average foraging distance of species s. P_{os} expresses the distance-weighted proportion of M parcels that are occupied by foraging pollinators and the relative abundance score of pollinators visiting each agricultural parcel. Finally, the floral resources were multiplied by the moving corridor of different species (100-500m). The result is a pollination services map, where nesting and foraging resources are reachable in the given moving corridor.

The structural diversity was evaluated by the Simpson Diversity Index (SIDI), the share of semi-natural habitat (SoSNH), and the number of the semi-natural habitat types (ToSNH).

Bailey et al. (2007) proposed the Simpson Diversity Index (*SIDI*) (Equation 15) for fine scale and heterogeneously defined biological groups, which was defined as:

$$SIDI = 1 - \left(\frac{\sum n(n-1)}{N(N-1)}\right) \qquad \textit{(Equation 15)}$$

where: n is the number of habitat patches and N is the total number of habitat types.

Billeter et al. (2008) found that the share of semi-natural habitat (SoSNH) and habitat diversity (ToSNH) correlated strongly with the species richness of several taxa. The share of semi-natural habitat was given by Equation 16:

$$SoSNH = \frac{SNH*100}{A} \qquad \textit{(Equation 16)}$$

Where: SNH wass the area in m^2 of semi-natural habitat types of the study site and A wass the size of the study site in m^2.

Habitat diversity (ToSNH) was the richness of the semi-natural habitat types in the LTS.

Table A2: List of provisioning and regulating ecosystem services (ES) according to CICES Classification (Haines-Young and Potschin, 2013) *linked to indicators addressed by agroforestry literature and methodologies to assess these indicators. SIDI: Simpson's diversity index, SoSNH: share of semi-natural habitat, ToSNH: Richness of semi-natural habitats.*

CICES Section - Division		**Ecosystem Service**	**ES Indicator**	**Methods and models**	**References**
Provisioning	Nutrition (Food / Feed)	Biomass production	1. Annual Biomass Yield 2. Biomass Stock	EcoYield-SAFE	van der Werf et al. 2007; Palma et al. submitted
	Material (Raw material, Genetic resources, Medicinal resources, Ornamental resources)				
	Energy				
	Water supply	Groundwater recharge	3. Groundwater recharge rate	Water balance equation using CropWat2.0 & MODIFFUS 3.0	Allen et al. 1998; Hürdler et al. 2015
Regulating and Maintenance	Regulation of biophysical environment (Air purification, Waste treatment)	Nutrient retention	4. Nitrate leaching (Phosphorus loss)	MODIFFUS 3.0	Hürdler et al. 2015
	Flow regulation (Disturbance prevention, regulation of water flows, erosion prevention)	Soil preservation	5. Erosion control	RUSLE	Renard et al. 1997; Panagos et al. 2015

Regulation of physicochemical environment (Climate regulation, Maintaining soil fertility)	Carbon Storage	6. Annual Carbon Sequestration 7. Carbon Stock	EcoYield-SAFE Yasso07	Liski et al. 2005; Palma et al. submitted
Regulation of biotic environment (Gene pool protection, Lifecycle maintenance, Pollination, Biological control)	Habitat and gene pool protection	8. Pollination Service 9. Flowering Resources 10. Cavity Nesting Resources 11. Ground Nesting Resources 12. SIDI 13. SoSNH 14. ToSNH	Pollination: Lonsdorf Habitat Richness: SIDI, SoSNH, ToSNH	Bailey et al. 2007; Billeter et al. 2008; Lonsdorf et al. 2009

Table A3: List of used datasets with title, provider and sources [ESDC: European Soil Data Centre, JRC: Joint research centre]

Dataset	Title	Provider	References	Resolution
Topography	Digital elevation model - SwissALTI3D	Swisstopo	(swisstopo, 2012)	2 m
Climate	Gempen(CH) 1960-2000	CliPick	(Palma, 2017)	
Soil	Total available water content (TAWC) for topsoil	ESDC, JRC	(Hiederer, 2013b, 2013a; Panagos et al., 2012)	1000 m
	Storing and filtering capacity		(Makó et al., 2017)	100 m
	Available water capacity		(Ballabio et al. 2016).	100 m
	Rainfall erosivity (R factor)		(Panagos et al., 2016)	1000 m
	Erodibility (K factor)		(Panagos et al., 2014)	500 m
	Groundcover (C factor)		(Panagos et al., 2015)	100 m
Water	CCM River and Catchment Database	ESDC, JRC	(Vogt et al., 2007)	100 m

Table A4: Land cover statistics of the landscape test sites (LTS) (F: forest, AF: agroforestry, A: agriculture, O: others).

LTS	Name	Municipality	Class	F	AF	A	O
				%			
AF1	Schönmatt	Gempen	AF	35	37	27	2
AF2	Wacht	Gempen	AF	37	55	2	6
AF3	Blauenstein	Seewen	AF	30	44	16	10
AF4	Güggelhof	Seewen	AF	32	31	19	19
NAF1	Ischlag	Gempen	NAF	28	0	64	7
NAF2	Nuglar	Nuglar - St. Pantaleon	NAF	28	0	62	8
NAF3	Rotenrain	Hochwald	NAF	39	0	58	3
NAF4	Ziegelschüren	Hochwald	NAF	26	0	71	3

Table A5: Summary of the outcomes summarized per LTS

Indicator	Unit	AF1	AF2	AF3	AF4	NAF1	NAF2	NAF3	NAF4
Annual Biomass Yield	t DM ha^{-1} yr^{-1}	5.4	4.4	4.7	4.1	6.3	7.1	6.3	5.9
Biomass Stock	t DM ha^{-1}	64.6	69.9	55.3	59.8	52.9	49.5	72.2	51.7
Groundwater Recharge Rate	%	42.5	42.3	46.5	47.0	48.4	48.6	47.7	51.4
Nitrate Leaching	kg N ha^{-1} yr^{-1}	10.5	5.4	8.9	5.8	14.0	16.4	12.8	12.1
Soil Erosion	t Soil ha^{-1} yr^{-1}	1.0	2.6	2.3	2.6	1.5	1.5	2.2	1.9
Annual Carbon Sequestration	t C ha^{-1} yr^{-1}	0.7	0.8	0.7	0.7	0.5	0.3	0.5	0.6
Carbon Stock	t C ha^{-1}	60.6	67.6	53.3	56.8	48.0	43.6	64.2	48.3
Flowering Resources	ha	21.5	24.7	22.8	18.2	17.7	11.6	11.2	16.8
Ground Nesting Resources	ha	46.3	89.2	64.7	65.4	33.7	34.2	58.9	58.0
Cavity Nesting Resources	ha	41.5	59.6	40.8	37.4	29.3	28.0	40.7	28.4
Pollination Services (Ground Nesting Species, 100m)	%	99.7	100	100.0	95.7	95.3	78.7	99.6	100
Pollination Services (Cavity Nesting Species, 100m)	%	99.7	100	96.5	94.5	92.5	69.4	85.2	91.7
Pollination Services (Ground Nesting Species, 350m)	%	100	100	100	100	100	100	100	100
Pollination Services (Cavity Nesting Species, 350m)	%	100	100	100	100	100	100	100	100
Pollination Services (Ground Nesting Species, 500m)	%	100	100	100	100	100	100	100	100
Pollination Services (Cavity Nesting Species, 500m)	%	100	100	100	100	100	100	100	100
SIDI		0.8	0.7	0.7	0.7	0.8	0.7	0.8	0.8
SoSNH	%	37	50	50	43	1	4	10	5
ToSNH	number	5	12	42	16	9	11	15	20

Annex II

I. According FADN (2017) the index is defined as:

[Total of output of crops and crop products, livestock and livestock products and of other output. Sales and use of (crop and livestock) products and livestock

+ change in stocks of products (crop and livestock)

+ change in valuation of livestock

- purchases of livestock

+ various non-exceptional products] /

[Specific costs

+ Overheads

+ Depreciation

+ External factors. These are costs linked to the agricultural activity of the holder and related to the output of the accounting year. Included are amounts relating to inputs produced on the holding (farm use) = seeds and seedlings and feed for grazing stock and granivores, but not manure. When calculating FADN standard results, farm taxes and other dues are not included in the total for costs but are taken into account in the balance "Subsidies and taxes" (subsidies - taxes) on current and non-current operations. The personal taxes of the holder are not to be recorded in the FADN accounts]

II. Database

Table A6: Monetary benefits and costs of ES indicators [in € ha^{-1} a^{-1}] of each landscape test side (LTS) of all case study regions.

Biogeographical region	Case study region	LTS ID	Biomass production (€ ha^{-1} a^{-1})	Groundwater recharge (€ ha^{-1} a^{-1})	Nutrient loss (€ ha^{-1} a^{-1})	Soil loss (€ ha^{-1} a^{-1})	Carbon storage (€ ha^{-1} a^{-1})	Pollination deficit (€ ha^{-1} a^{-1})	NET ESvalue (€ ha^{-1} a^{-1})
Atlantic	ES3	NAF1	7.49	0.82	-23.93	-37.52	0.77	0	-52.37
Atlantic	ES3	NAF2	6.79	0.84	-20.4	-11.54	2.75	0	-21.55
Atlantic	ES3	NAF3	10.56	0.86	-43.18	-16.76	1.26	0	-47.26
Atlantic	ES3	NAF4	9.77	0.92	-46.2	-32	0.68	0	-66.84
Atlantic	ES3	AF1	17.05	0.72	-7.73	-2.22	4.14	0	11.96
Atlantic	ES3	AF2	41.09	0.8	-33.7	-16.94	1.19	0	-7.57
Atlantic	ES3	AF3	23.6	0.74	-10.5	-13.04	4.75	0	5.54
Atlantic	ES3	AF4	13.92	0.84	-23.79	-31.05	1.91	0	-38.17
Atlantic	FR	NAF1	68.56	0.46	-127.82	-9.62	0.09	0	-68.33

Atlantic	FR	NAF2	60.57	0.46	-108.3	-19.3	0.32	0	-66.25
Atlantic	FR	NAF3	106.78	0.42	-138.25	-21.41	0.21	0	-52.25
Atlantic	FR	NAF4	65.66	0.43	-116.24	-15.88	0.27	0	-65.76
Atlantic	FR	AF1	47.34	0.43	-104.04	-6.63	0.73	0	-62.17
Atlantic	FR	AF2	62.08	0.43	-114.16	-24.49	1.04	0	-75.1
Atlantic	FR	AF3	43.58	0.41	-85.8	-16.75	1.06	0	-57.49
Atlantic	FR	AF4	44.98	0.41	-104.43	-12.34	0.53	0	-70.84
Atlantic	UK	NAF1	39.53	0.32	-95.99	-1.71	0.61	0	-57.24
Atlantic	UK	NAF2	45.87	0.33	-110.5	-1.61	1.74	0	-64.17
Atlantic	UK	NAF3	5.89	0.09	-11.86	-0.03	4.29	0	-1.63
Atlantic	UK	NAF4	60.97	0.3	-108.86	-0.27	0	0	-47.85
Atlantic	UK	AF1	7.16	0.19	-13.13	-1.95	2.46	0	-5.26
Atlantic	UK	AF2	3.73	0.2	-20.32	-0.17	3.99	0	-12.57
Atlantic	UK	AF3	5.07	0.13	-14.06	-0.12	3.44	0	-5.54
Atlantic	UK	AF4	67.6	0.3	-158.39	-0.24	1.81	0	-88.92
Continental	CH1	NAF1	38.12	0.83	-55.91	-9.9	2.53	0	-24.33
Continental	CH1	NAF2	56.1	0.81	-65.57	-9.78	1.63	0	-16.8
Continental	CH1	NAF3	37.2	0.83	-51.02	-14.47	2.65	0	-24.81
Continental	CH1	NAF4	36.39	0.9	-48.26	-12.28	3.07	0	-20.18
Continental	CH1	AF1	25.26	0.68	-41.99	-6.24	3.58	0	-18.72
Continental	CH1	AF2	17.6	0.67	-21.64	-16.61	4.15	0	-15.82
Continental	CH1	AF3	18.26	0.78	-35.51	-14.75	3.7	0	-27.52
Continental	CH1	AF4	18.25	0.8	-23.13	-16.93	3.56	0	-17.45
Continental	CH2	NAF1	11.71	0.68	-35.03	-13.21	4.21	0	-31.64
Continental	CH2	NAF2	7.25	0.76	-25.11	-18.87	4.65	0	-31.31
Continental	CH2	NAF3	8.78	0.7	-31.1	-9.88	2.88	0	-28.61
Continental	CH2	NAF4	16.89	0.7	-38.37	-37.7	1.28	0	-57.2
Continental	CH2	AF1	10.23	0.67	-24.62	-28.58	5.33	0	-36.99
Continental	CH2	AF2	8.8	0.66	-18.89	-5.92	5.8	0	-9.54
Continental	CH2	AF3	9.24	0.7	-23.64	-9.08	6.09	0	-16.68
Continental	CH2	AF4	7.58	0.7	-26.67	-15.71	5.31	0	-28.8
Continental	DE	NAF1	36.46	0.55	-118.33	-12.85	0.1	-1	-95.07
Continental	DE	NAF2	45.03	0.61	-88.82	-10.65	0.02	-11	-64.81
Continental	DE	NAF3	68.41	0.54	-94.91	-9.75	0.12	0	-35.59
Continental	DE	NAF4	34.55	0.59	-111.83	-5.52	0.08	-1	-83.12
Continental	DE	AF1	36.72	0.49	-99.9	-7.86	0.49	0	-70.06
Continental	DE	AF2	30.96	0.5	-99.3	-8.12	0.66	0	-75.31
Continental	DE	AF3	32.99	0.58	-88.68	-10.43	0.77	-7	-71.77
Continental	DE	AF4	30.13	0.49	-77.33	-5.1	1.04	0	-50.77
Continental	RO	NAF1	12.06	0.09	-34.51	-9.81	1.94	0	-30.23
Continental	RO	NAF2	34.8	0.08	-26.44	-8.38	3.21	0	3.26
Continental	RO	NAF3	4.38	0.09	-35.8	-14.54	1.35	-2	-46.52
Continental	RO	NAF4	3.92	0.09	-30.2	-11.18	1.47	0	-35.9
Continental	RO	AF1	57.13	0.06	-19.7	-6.03	4.86	0	36.32
Continental	RO	AF2	26.76	0.07	-23.13	-7.65	3.32	0	-0.62
Continental	RO	AF3	28.5	0.08	-26.48	-6.45	3.23	0	-1.12

Continental	RO	AF4	35.31	0.07	-24.97	-7.07	3.58	0	6.92
Mediterranean	ES1	NAF1	2.18	0.27	-23.8	-5.37	1.49	0	-25.23
Mediterranean	ES1	NAF2	2.18	0.27	-23.51	-5.48	1.49	0	-25.06
Mediterranean	ES1	NAF3	3.02	0.26	-21.18	-4.53	1.55	0	-20.88
Mediterranean	ES1	NAF4	2.18	0.27	-19.99	-3.14	1.49	0	-19.19
Mediterranean	ES1	AF1	12.65	0.25	-6.09	-3.27	1.57	0	5.12
Mediterranean	ES1	AF2	4.01	0.26	-1.65	-5.21	1.57	0	-1.03
Mediterranean	ES1	AF3	10.03	0.25	-2.98	-3.81	1.49	0	4.98
Mediterranean	ES1	AF4	49.45	0.27	-21.41	-6.58	1.16	0	22.88
Mediterranean	ES2	NAF1	90.06	0.3	-71.49	-30.67	0.38	0	-11.42
Mediterranean	ES2	NAF2	91.39	0.3	-58.44	-26.81	0.57	0	7
Mediterranean	ES2	NAF3	19.49	0.27	-60.15	-19.12	0.13	0	-59.38
Mediterranean	ES2	NAF4	50.89	0.29	-77.24	-23.51	0.2	0	-49.38
Mediterranean	ES2	AF1	2.38	0.13	-11.56	-0.1	1.53	0	-7.62
Mediterranean	ES2	AF2	74.37	0.21	-30.47	-9.75	1.1	0	35.46
Mediterranean	ES2	AF3	23.57	0.16	-18.17	-5.11	1.59	0	2.04
Mediterranean	ES2	AF4	14.93	0.16	-16.11	-2.75	2.16	0	-1.61
Mediterranean	GR	NAF1	157.99	0.08	39.49	3.83	0.24	-68	133.62
Mediterranean	GR	NAF2	119.01	0.1	49.19	5.5	0.11	-4	169.9
Mediterranean	GR	NAF3	115.98	0.09	46.02	4.77	0.1	-3	163.97
Mediterranean	GR	NAF4	136.28	0.1	49.45	5.25	0.14	-9	182.21
Mediterranean	GR	AF1	122.45	0.09	29.3	11.23	1.34	0	164.4
Mediterranean	GR	AF2	71.17	0.06	12.32	10.95	1.69	0	96.2
Mediterranean	GR	AF3	99.17	0.07	18.98	12.68	1.99	0	132.89
Mediterranean	GR	AF4	110.79	0.08	24.51	19.22	1.79	0	156.39
Mediterranean	PT	NAF1	9.64	0.32	-30.4	-4.25	1.54	0	-23.16
Mediterranean	PT	NAF2	17.11	0.31	-27.73	-2.14	1.5	0	-10.95
Mediterranean	PT	NAF3	3.23	0.33	-35.38	-2.79	1.38	0	-33.24
Mediterranean	PT	NAF4	43.55	0.3	-29.21	-4.9	1.52	0	11.26
Mediterranean	PT	AF1	108.28	0.16	-12.39	-0.5	1.76	0	97.32
Mediterranean	PT	AF2	76.55	0.17	-11.9	-0.1	1.65	0	66.37
Mediterranean	PT	AF3	164.67	0.17	-12.24	-0.1	1.72	0	154.23
Mediterranean	PT	AF4	109.16	0.16	-12.42	-0.2	1.64	0	98.35

Annex III

Tables A1 – A4:Proposed agroforestry systems clustered by biogeographical region and agroforestry type showing system description, tree biomass production (based on indicated literature and IPCC (1997) for root biomass assessment) and carbon storage potential (based on indicated literature or – if unknown (Aalde et al., 2006) *– assuming 50% of tree biomass to be carbon). SRC: Short rotation coppice.*

A1: Atlantic Agroforestry practices

ID	Biogeo-graphical region	AF type	Title	Tree / hedgerow species	Trees [trees ha^{-1}], hedgerow [m ha^{-1}] or wood cover [% ha^{-1}]	Planting and management system	Crop species and products	Tree products	Year of tree harvesting	Tree and root biomass, references [t $ha^{-1}a^{-1}$]		Carbon storage, references [t C $ha^{-1}a^{-1}$]	
1	Atlantic grassland	silvopastoral, hedgerows	Field edges planted with hedgerows	common ash (*Fraxinus excelsior*), mountain ash (*Sorbus aucuparia*), hornbeam (*Carpinus betulus*), hazel (*Corylus avelana*)	288 trees ha^{-1} (8% ha^{-1})	at the edges	grazing, hay, silage	woodchips	3 – 5 years	0.4 - 1.94	Case study UK, France (Kay et al., 2018b)	0.2 - 0.95	Case study UK, France (Kay et al., 2018b)
2	Atlantic grassland	silvopastoral, coppice	SRC agroforestry for ruminants	willow (Salix spp), alder (Alnus glutinosa)	0.25 /0.7 m x 24m (34,4% ha^{-1})	lines	grazing, hay, silage	fodder-trees, woodchips	5 – 8 years	1.02 - 2.97	(Bärwolff et al., 2012)	0.51 - 1.48	(Aalde et al., 2006)
3	Atlantic grassland	silvopastoral, coppice	Fodder and energy trees	willow (*Salix viminalis*), poplar (*Populus* sp.), hazel (*Corylus avellana*), alder (*Alnus glutinosa*)	1175 trees ha^{-1} (0.7- 1.0 m within rows, 24m within twin rows, 34% ha^{-1})	lines (twin lines)	grazing, hay, silage	fodder-trees, woodchips	15 years	1.02 - 2.97	(Bärwolff et al., 2012)	0.51 - 1.48	(Aalde et al., 2006)
4	Atlantic grassland	silvopastoral, coppice	Agroforestry for ruminants in France	pear (*Pyrus* spp), honey locust (*Gleditsia triacanthos*), service tree (*Sorbus domestica*), white mulberry (*Morus alba*), Italian alder (*Alnus cordata*), goat willow (*Salix caprea*), field elm (*Ulmus minor*), black locust (*Robinia pseudoacacia*), grey alder (*Alnus incana*)	(single -2 m, double -6 m triple -10 m), 4 m trees, 1.3m coppices x 20m, (11% ha^{-1})	(single, double, triple) lines	grazing, hay, silage	fodder-trees, woodchips	5 – 8 years	0.33 - 0.96	(Bärwolff et al., 2012)	0.16 - 0.48	(Aalde et al., 2006)
5	Atlantic grassland	silvopastoral, coppice	SRC, fodder trees	Pedunculate oak (*Quercus robur*), sycamore (*Platanus occidentalis*), cherry (*Prunus avium*)	6 x 1.5 m (1056 trees ha^{-1} or 64% ha^{-1}), 8 x 1.5 m (726 trees ha^{-1}, 44 % ha^{-1})	lines	grazing, hay, silage	woodchips	5 – 8 years	1.31 - 5.6	(Lawson et al., 2016); (Bärwolff et al., 2012)	0.66 - 2.8	(Aalde et al., 2006)
6	Atlantic grassland	silvopastoral, single trees	High stem timber trees	poplar (*Populus* spp)	25 trees ha^{-1} (5% ha^{-1})	boundary	grazing, hay, silage	timber	25 years	0.9 - 2.06	(Graves et al., 2010); (Unseld, 2017)	0.46 - 1.05	(Fang et al., 2010)
7	Atlantic grassland	silvopastoral, single trees	High stem timber trees	poplar (*Populus* spp)	400 trees ha^{-1}, after 15-20 years: 120-150 trees ha^{-1}	lines	grazing, hay, silage	timber	first cut: 15-20 years, harvest: 25-30 years	5.41 - 12.38	(Lawson et al., 2016); (Graves et al., 2010)	2.78-6.35	(Fang et al., 2010)

ID	Biogeo-graphical region	AF type	Title	Tree / hedgerow species	Trees [trees ha^{-1}], hedgerow [m ha^{-1}] or wood cover [% ha^{-1}]	Planting and management system	Crop species and products	Tree products	Year of tree harvesting	Tree and root biomass, references [t $ha^{-1}a^{-1}$]		Carbon storage, references [t C $ha^{-1}a^{-1}$]	
8	Atlantic grassland	silvopastoral, single trees	High stem forest trees	common ash (*Fraxinus excelsior*), Pedunculate oak (*Quercus robur*)	5 x 5 m (400 trees ha^{-1})	single tree scattered	grazing, hay, silage	timber	15 years	1.38 - 2.63	(Lawson et al., 2016); British, German forest tables	0.69-1.31	(Aalde et al., 2006)
9	Atlantic grassland	silvopastoral, single trees	Fruit and fodder trees	walnut (*Juglans regia*), Pedunculate oak (*Quercus robur*) (including edible acorns – *Acer campestre*), sweet chestnut (*Castanea sativa*), cider apple trees (*Malus domestica*)	400 and 1,000 trees ha^{-1}	lines	grazing, hay, silage	fruits, fodder-trees, woodchips, timber	15 years	walnut: 4.87-7.79, oak: 0.86-2.14	(Lawson et al., 2016); British, German forest tables	walnut: 2.92 - 4.68, oak: 0.43-1.07	(Cardinael et al., 2017) (Aalde et al., 2006)
10	Atlantic grassland	silvopastoral, single trees	High stem timber trees	pawlonia (*Paulownia tomentosa*), dutch elm (*Ulmus × hollandica*)	8 x 1.5 m (726 trees ha^{-1}, 44 % ha^{-1})	lines	grazing, hay, silage	timber	15 years	1.17 - 3.85	(Woods, 2008); (Durán Zuazo et al., 2013); (García-Morote et al., 2014); (Lawson et al., 2016)	0.58 - 1.93	(Aalde et al., 2006)
11	Atlantic grassland	silvopastoral, single trees	Traditional orchard	fruit trees (apple – *Malus domcestica*, pear – *Pyrus* spp, plum – *Prunus domestica*)	80 trees ha^{-1}	lines	grazing, hay, silage	fruits (woodchips)	60 years	2.33	(Schnitzler et al., 2014); (Lawson et al., 2016)	1.23	(Johnson and Gerhold, 2001)
12	Atlantic grassland	silvopastoral, single trees	Fruit trees	apples (*Malus domestica*), pears (*Pyrus communis*), plums (*Prunus domestica*), cherries (*Prunus avium*) and other fruit and nuts	650-750 trees ha^{-1} (3.5-4.5 m x 2-2.5 m)	lines	grazing	fruits, nuts, woodchips	12 - 15 years	10.6	(Winzer et al., 2017)	5.3	(Aalde et al., 2006)
13	Atlantic grassland	silvopastoral, single trees	High stem fodder trees	common ash (*Fraxinus excelsior*)	400 trees ha^{-1} (two thinnings then 120-150 trees ha^{-1}, 5 x 5m)	lines	grazing (ryegrass)	fodder-trees, woodchips	first cut: 15-20 years, harvest: 25-30 years	1.03-1.97	British, German forest tables	0.51-0.98	(Aalde et al., 2006)
14	Atlantic grassland	silvopastoral, single trees	Fodder trees	broadleaf species, e.g. Pedunculate oak (*Quercus robur*), sycamore (*Platanus occidentalis*), cherry (*Prunus avium*), beech (*Fagus sylvatica*)	200-400 trees ha^{-1}. Must maintain initial planting density	lines	grazing	fodder-trees, woodchips	Land must be available for grazing for at least 20 years	1.03-1.97	British, German forest tables	0.51-0.98	(Aalde et al., 2006)
15	Atlantic arable	silvoarable, hedgerows	Bocage	mixed hedgerow - species:field maple (*Acer campestre*), common birch (*Betula pendula*), apple (*Malus domestica*), cherries (*Prunus avium*), *Sorbus* spp, *Quercus* spp	5 - 8% ha^{-1}	boundery	cereals (wheat, barley, oats),	woodchips, timber, delimitation of properties, shelter	every 15 years	0.4 - 1.94	Case study UK, France (Kay et al., 2018b)	0.2 - 0.95	Case study UK, France (Kay et al., 2018b)

ID	Biogeo-graphical region	AF type	Title	Tree / hedgerow species	Trees [trees ha^{-1}], hedgerow [m ha^{-1}] or wood cover [% ha^{-1}]	Planting and management system	Crop species and products	Tree products	Year of tree harvesting	Tree and root biomass, references [t $ha^{-1}a^{-1}$]		Carbon storage, references [t C $ha^{-1}a^{-1}$]	
16	Atlantic arable	silvoarable, hedgerows	Productive boundary hedgerow	mixed hedgerow species: hawthorn (*Crataegus* spp), blackthorn (*Prunus spinosa*), field maple (*Acer campestre*), hazel (*Corylus avellane*)	0.03% ha^{-1}	boundary hedgerow	crop rotation with cereals (wheat, barley, oats), potatoes, squash, organic fertility building ley	woodchip	every 15 years	0.2 - 0.95	Case study UK, France (Kay et al., 2018b)	0.1 - 0.45	Case study UK, France (Kay et al., 2018b)
17	Atlantic arable	silvoarable, coppies	SRC	hornbeam (*Carpinus betulus*), common ash (*Fraxinus excelsior*), alder (*Alnus cordata*)	572 trees ha^{-1} (11% ha^{-1})	lines	crop rotation, multiple crops	woodchips	4 - 6 years	0.33 - 0.96	(Bärwolff et al., 2012)	0.16 - 0.48	(Aalde et al., 2006)
18	Atlantic arable	silvoarable, coppice	SRC	poplar (*Populus* spp), willow (*Salix viminalis*)	18% ha^{-1} (48m cropping)	lines	crop rotation (wheat, oilseed, barley)	woodchips	5 -8 years	0.54 - 1.57	(Bärwolff et al., 2012)	0.27-0.78	(Aalde et al., 2006)
19	Atlantic arable	silvoarable, coppice	Alley cropping - SRC	willow (*Salix viminalis*), hazel (*Corylus avellana*)	1000-1300 trees ha^{-1} (24% ha^{-1})	twin rows with 10-15m wide crop alley	cereals (wheat, barley, oats), potatoes, squash, organic fertility building ley	woodchips	every 2 years for willow, every 5 years for hazel	0.72 - 2.1	(Bärwolff et al., 2012)	0.36-1.05	(Aalde et al., 2006)
20	Atlantic arable	silvoarable, single trees	High stem Walnut	walnut (*Juglans intermedia*)	48 -50 trees ha^{-1} (5% ha^{-1})	lines	crop rotation multiples	timber	60 years	0.97 - 2.08	(Sereke et al., 2015); (Cardinael et al., 2017)	0.58 - 1.25	(Cardinael et al., 2017)
21	Atlantic arable	silvoarable, single trees	High stem timber trees	walnuts (*Juglans regia*), maples (*Acer* spp), wild cherry (*Prunus avium*), checker tree, (*Sorbus torminalis*), service tree (*Sorbus domestica*), apple (*Malus domestica*), pear (*Pyrus* spp).	28-110 trees ha^{-1}, (26-50 m between rows)	lines		timber	60 years	walnut: 0.54 - 4.58, cherry: 0.35 - 2.61	German forest tables, (Sereke et al., 2015); (Cardinael et al., 2017)	walnut: 0.32 - 2.75, cherry: 0.19 - 1.4	(Cardinael et al., 2017)
22	Atlantic arable	silvoarable, single trees	Alley cropping	mixed hardwood: lime (*Tilia cordata*), hornbeam (*Carpinus betulus*), cherry (*Prunus avium*), alder (*Alnus cordata*), common ash (*Fraxinus excelsior*), maple (*Acer pseudoplatanus*), sessil oak (*Quercus petraea*)	150 trees ha^{-1}	twin rows with 10-15m wide crop alley	cereals (wheat, barley, oats), potatoes, squash, organic fertility building ley	timber, woodchips	harvesting depends on species, estimated from 25 years to 100 years. Pollarding on selected species every 5-10 years	0.32 - 1.93	British forest tables	0.16 - 0.51	(Aalde et al., 2006)
23	Atlantic arable	silvoarable, single trees	Alley cropping	fruit trees: apple (*Malus domestica*), pear (*Pyrus* spp), plum (*Prunus domestica*)	85-100 trees ha^{-1}	single rows with 24m wide crop alley	cereals and organic fertility building ley	fruits (timber)	fruit harvested annually	apple: 2.47-2.91	(Schnitzler et al., 2014)	apple: 1.31-1.54	(Johnson and Gerhold, 2001)

A2: Continental Agroforestry practices

ID	Biogeographical region	AF type	Title -	Tree / hedgerow species	Trees [trees ha^{-1}], hedgerow [m ha^{-1}] or wood cover [% ha^{-1}]	Planting and management system	Crop species and products	Tree products	Year of tree harvesting	Tree and root biomass, references [t $ha^{-1}a^{-1}$]		Carbon storage, references [t C $ha^{-1}a^{-1}$]	
24	Continental lowlands, grassland	silvopastoral, coppice	Agroforestry for free-range pig production	poplar (*Populus* spp), willow (*Salix* spp), various fruit trees	10-40% ha^{-1}, (2,5x3,5m)	SRC lines	grazing, hay, silage	woodchips, fodder-trees	5-8 years	poplar: 0.86-2.75	(Mirck et al., 2016)	poplar: 0.44-1.41	(Fang et al., 2010)
25	Continental lowlands, arable	silvoarable, coppice	Alley cropping	poplar (*Poplus* spp); black locust (*Robinia pseudoacacia*)	single rows: 9700 trees ha^{-1}, Twin rows: 8700 trees ha^{-1} (12.5% ha^{-1}).	single and twin rows with 24m, 48m, and 96m wide crop alleys.	crop rotation (wheat, maize, oilseed rape, barley)	woodchips	poplar: every 3-4 years; robiniat: every 4-5 years.	poplar: 0.86 robinia: 1.08	(Mirck et al., 2016)	poplar: 0.44, robinia: 0.54	(Fang et al., 2010)
26	Continental lowlands, arable	silvoarable, coppice	Alley cropping	black locust (*Robinia pseudoacacia*)	twin rows: 9200 trees ha^{-1} (34.4% ha^{-1}).	twin rows with 24m wide crop alleys.	crop rotation (wheat, maize, oilseed rape, barley)	woodchips	every 3-6 years.	2.02	(Kanzler et al., 2014)	1.01	(Aalde et al., 2006)
27	Continental lowlands, arable	silvoarable, coppice	Alley cropping	poplar (*Poplus* spp); Mixed hedgerow species: willow (*Salix* spp), hornbeam (*Carpinus betulus*), common ash (*Fraxinus excelsior*), common birch (*Betula pendula*), black locust (*Robinia pseudoacacia*)	rows A, B, and C: 10'000 trees ha^{-1}, rows D, E, F, and G: 2222 trees ha^{-1}, (10% ha^{-1}).	single and twin rows with 48m, 96m, and 144m wide crop alleys.	crop rotation (wheat, maize, oilseed rape, barley)	woodchips	rows A, B, and C: every 3-5 years. rows D, E, F, and G: every 8 – 10 years.	0.3 - 0.88	(Bärwolff et al., 2012)	0.15 - 0.44	(Fang et al., 2010)
28	Continental lowlands, arable	silvoarable, single trees	Mixed timber and wild fruit species	Grayish oak (*Quercus pedunculiflora*), field maple (*Acer campestre*), lime (*Tilia* spp), hawthorn (*Crataegus* sp), *Rosa* spp, blackthorn (*Prunus spinosa*)	100 trees ha^{-1}	lines	vegetable	fruits, fodder-trees, timber	harvesting depends on species estimated from 25 years to 120 years.	oak: 3.11; tilia: 2.65	(Constandache et al., 2012, 2006; Costăchescu et al., 2012; Dănescu et al., 2007), Hungarian, German forest tables	oak: 1.59, tilia:1.32	(Aalde et al., 2006)
29	Continental hills, grassland	silvopastoral, single trees	Wooded grassland	sessil oak (*Quercus petraea*), beech (*fagus sylvatica*), hornbeam (*Carpinus betulus*), wild fruit trees; mixed poplar (*Poplus* spp.), willow (*Salix* spp.)	50-300 trees ha^{-1} (~10-50% ha^{-1})	scattered	grazing, hay, silage	acorns, fruits, timber, (fodder-trees)	not harvested	oak: 1.38-5.51, beech: 1.18-4.74, hornbeam: 0.77- 3.11	(Roellig et al., 2018), Hungarian, Czech forest tables	oak: 0.71 - 2.83, beech: 0.59- 2.34, hornbeam: 0.38 - 1.55	(Aalde et al., 2006)

ID	Biogeographical region	AF type	Title -	Tree / hedgerow species	Trees [trees ha^{-1}], hedgerow [m ha^{-1}] or wood cover [% ha^{-1}]	Planting and management system	Crop species and products	Tree products	Year of tree harvesting	Tree and root biomass, references [t $ha^{-1}a^{-1}$]		Carbon storage, references [t C $ha^{-1}a^{-1}$]	
30	Continental hills, grassland	silvopastoral, single trees	Grazed orchard	classical standard fruit trees: cherry (*Prunus avium*), walnut (*Juglans regia*), apple (*Malus domestica*)	100 – 120 trees ha^{-1}	lines	grazing, hay, silage	fruits, timber	70 - 90 years	cherry: 1.28 - 2.85, walnut: 2.39-3.87	German, Hungarian forest tables, (Sereke et al., 2015)	cherry: 0.69-1.53, walnut: 1.43-2.32	(Cardinael et al., 2017)
31	Continental hills, grassland	silvopastoral, single trees	Grazed orchard	wild fruit varieties for valuable wood (e.g cherry - *Prunus avium*)	100 – 120 trees ha^{-1}	lines	grazing, hay, silage	timber	80 - 120 years	cherry: 1.28 - 2.85	German forest tables, (Sereke et al., 2015)	cherry: 0.69-1.53	(Cardinael et al., 2017)
32	Continental hills, grassland	silvopastoral, single trees	Wooded grassland	fruit trees: cherry (*Prunus avium*), walnut (*Juglans regia*), apple (*Malus domestica*), etc.	60 trees ha^{-1}	lines	grazing, hay, silage	fruits	70-90 years	cherry: 0.76 - 1.42, apple: 1.75-2.71, walnut: 1.43-1.93	German, Hungarian forest tables, (Schnitzler et al., 2014); (Sereke et al., 2015)	cherry: 0.41-0.76, apple: 0.93-1.43, walnut: 0.86 -1.16	(Johnson and Gerhold, 2001); (Cardinael et al., 2017)
33	Continental hills, arable	silvoarable, coppice	SRC	willow (*Salix* spp)	18% ha^{-1} (48m cropping)	lines	crop rotation (wheat, maize, oilseed rape, barley)	woodchips	5 - 8 years	0.54 - 1.57	(Bärwolff et al., 2012)	0.27-0.78	(Aalde et al., 2006)
34	Continental hills, arable	silvoarable, single trees	Intercropped high stem fruit trees	old, robust apple varieties, (*Malus* e.g. Bohnapfel, Boskoop, Schneiderapfel, Glockenapfel, etc.)	60 – 70 trees ha^{-1}	lines	intensive special crop cultivation, vegetable growing, herb cultivation, berry cultivation	fruits, timber	70 - 90 years	apple: 1.75-2.71	(Schnitzler et al., 2014)	apple: 0.93-1.43	(Johnson and Gerhold, 2001)
35	Continental hills, arable	silvoarable, single trees	Intercropped wild fruit varieties and nut trees	nut trees and wild fruit varieties e.g wild cherry (*Prunus avium*), service tree (*Sorbus* sp.), mulberry tree (*Morus* sp.)	50 trees ha^{-1}	lines	intensive cultivation of arable crops	timber	80 - 120 years	cherry: 0.64 - 1.18, walnut: 1.19-1.61	German, Hungarian forest tables, (Sereke et al., 2015)	cherry: 0.34-0.64, walnut: 0.71-0.96	(Cardinael et al., 2017)
36	Continental hills, arable	silvoarable, single trees	Orchard with vegetables or fruits (strawberries)	fruit trees: cherry (*Prunus avium*), walnut (*Juglans regia*), apple (*Malus domestica*), etc.	60 trees ha^{-1}	lines	vegetable, berries (strawberries)	fruits, timber	70-90 years	cherry: 0.76 - 1.42, apple: 1.75-2.71, walnut: 1.43-1.93	German, Hungarian forest tables, (Schnitzler et al., 2014); (Sereke et al., 2015)	cherry: 0.41-0.76,apple: 0.93-1.43, walnut: 0.86 -1.16	(Johnson and Gerhold, 2001); (Cardinael et al., 2017)
37	Continental hills, arable	silvoarable, single trees	Paulownia / alfalfa	pawlonia (*Paulownia tomentosa*)	126 trees ha^{-1} (18 m x 5 m)	lines	triticale, alfalfa	timber	10-12 years	7.54	(Stimm et al., 2013); (Vityi et al., 2016)	3.77	(Aalde et al., 2006)

ID	Biogeographical region	AF type	Title -	Tree / hedgerow species	Trees [trees ha^{-1}], hedgerow [m ha^{-1}] or wood cover [% ha^{-1}]	Planting and management system	Crop species and products	Tree products	Year of tree harvesting	Tree and root biomass, references [t $ha^{-1}a^{-1}$]		Carbon storage, references [t C $ha^{-1}a^{-1}$]	
38	Continental hills, arable	silvoarable, single trees	Mixed timber and wild fruit species	grayish oak (*Quercus pedunculiflora*), field maple (*Acer campestre*), lime (*Tilia* sp.), hawthorn (*Crataegus* sp), *Rosa* sp, blackthorn (*Prunus spinosa*)	100 trees ha^{-1}	lines	vegetable, berries (strawberries)	fruits, flowers, fodder-trees	Harvesting depends on species estimated from 25 years to 120 years.	oak: 3.11; tilia: 2.65	(Constandache et al., 2012, 2006; Costăchescu et al., 2012; Dănescu et al., 2007), Hungarian, German forest tables	oak: 1.59, tilia: 1.32	(Aalde et al., 2006)
39	Continental hills, arable	silvoarable, single trees	Intercropped high stem fruit trees	modern, resistant fruit varieties for high-stem fruit trees apple varieties (*Malus* e.g. Topaz, Re-varieties, Spartan, Ariwa, Rowina, Golden)	50 trees ha^{-1}	lines	crop rotation wheat, oilseed rape, spelt, field-peas, sunflower	fruits, timber	70 - 90 years	apple: 1.45-2.25	(Schnitzler et al., 2014)	apple:0.77-1.19	(Johnson and Gerhold, 2001)

A3: Mediterranean Agroforestry practices

ID	Biogeographical region	AF type	Title	Tree / hedgerow species	Trees [trees ha^{-1}], hedgerow [m ha^{-1}] or wood cover [% ha^{-1}]	Planting and management system	Crop species and products	Tree products	Year of tree harvesting	Tree and root biomass, references [t $ha^{-1}a^{-1}$]		Carbon storage, references [t C $ha^{-1}a^{-1}$]	
40	Mediterranean lowlands, arable	silvoarable, single trees	High stem timber trees	hybrid poplar (*Populus* spp); Pedunculate oak (*Quercus robur*)	57 trees ha^{-1}	lines	cereals	timber	poplar: 15 years; oak: 35 years	4.0 - 4.9	Italian, Spanish forest tables	2.08-2.55	(Fang et al., 2010)
41	Mediterranean lowlands, arable	silvoarable, single trees	High stem timber trees	pedunculate oak (*Quercus robur*)	57 trees ha^{-1}	lines	cereals	timber	35 years	0.2 - 0.52	Italian, French forest tables	0.11 -0.26	(Aalde et al., 2006)
42	Mediterranean lowlands, arable	silvoarable, coppice	SRC	pawlonia (*Paulownia tomentosa*)	500 trees ha^{-1}	lines	crop rotation barley, wheat, peas	woodchips	every 2-3 years	0.8 - 4.5	(Durán Zuazo et al., 2013); (Stimm et al., 2013); (García-Morote et al., 2014)	0.4 - 2.2	(Aalde et al., 2006)
43	Mediterranean lowlands, arable	silvoarable, single trees	Timber plantation	pawlonia (*Paulownia tomentosa*)	200 trees ha^{-1}	lines	crop rotation wheat, sunflower, peas	fodder-trees, timber	12 years	10.0 - 12.0	(Stimm et al., 2013)	5.0 -6.0	(Aalde et al., 2006)

ID	Biogeographical region	AF type	Title	Tree / hedgerow species	Trees [trees ha^{-1}], hedgerow [m ha^{-1}] or wood cover [% ha^{-1}]	Planting and management system	Crop species and products	Tree products	Year of tree harvesting	Tree and root biomass, references [t $ha^{-1}a^{-1}$]		Carbon storage, references [t C $ha^{-1}a^{-1}$]	
44	Mediterranean hills, grassland	silvopastoral, single trees	Grazed fruit plantations	olive (*Olea europaea*), almond (*Prunus dulcis*)	250 trees ha^{-1}	lines	grazing, legume rich mix (annual self seeding species)	fruits, oil, nuts	annual prunings, not harvested	olive: 3.39, almond:2.71	(Spinelli and Picchi, 2010); (Velázquez-Martí et al., 2011)	olive: 1.97, almond:1.36	(Proietti et al., 2014); (Lopez-Bellido et al., 2016)
45	Mediterranean hills, grassland	silvopastoral, single trees	Dehesa	holm oak (*Quercus ilex*)	25-50 trees ha^{-1}	scattered	grazing	acorns, fodder-trees	not harvested	0.19 – 0.31	(Palma et al., 2017)	0.09 – 0.16	Case study Spain (Kay et al., 2018b)
46	Mediterranean hills, grassland	silvopastoral, single trees	Grazed cork oak plantation	cork oak (*Quercus suber*)	113 trees ha^{-1}, after 20 years: 50 trees ha^{-1}	lines	grazing	cork, timber	80 years	1.46 - 4.29	(Palma et al., 2014)	0.34-1.29	(Palma et al., 2014)
47	Mediterranean hills, arable	silvoarable, single trees	Fruit plantations	fruit trees: apple (*Malus domestica*), pear (*Pyrus* spp.), cherry (*Prunus avium*), etc.	417 trees ha^{-1}	lines	fodder crops (alfalfa)	fruits	not harvested	10.60	(Winzer et al., 2017)	5.3	(Aalde et al., 2006)
48	Mediterranean hills, arable	silvoarable, single trees	Fruit plantations	olive (*Olea europaea*), almond (*Prunus dulcis*)	250 trees ha^{-1}	lines	crop rotation barley, arley, fallow	fruits, oil, nuts	annual prunings, not harvested	olive: 3.39, almond: 2.71	(Spinelli and Picchi, 2010); (Velázquez-Martí et al., 2011)	olive: 1.97, almond:1.36	(Proietti et al., 2014); (Lopez-Bellido et al., 2016)
49	Mediterranean hills, arable	silvoarable, single trees	Durum wheat production in agroforestry	poplar (*Populus* spp), walnut (*Juglans nigra x regia*), plum (*Prunus domestica*), common ash (*Fraxinus excelsior*), maple (*Acer* spp), hackberry (*Celtis australis*), wild pear tree (*Pyrus pyraster*)	128 trees ha^{-1} (6x13m)	lines	durum wheat	timber	poplar: 15-20 years; walnut: 35-50 years	poplar: 7.19 - 9.81, walnut: 2.49 - 5.33	(Cardinael et al., 2017), Italian forest tables	poplar: 3.69-4.67; walnut: 0.77 - 1.85	(Fang et al., 2010); (Cardinael et al., 2017)
50	Mediterranean hills, arable	silvoarable, single trees	Walnut trees intercropped with wheat	hybrid walnut (*Juglans nigra x regia*)	104 trees ha^{-1} (8 x13m)	lines	wheat	timber	40-60 years	1.05 - 4.33	(Moreno et al., 2016a) ; (Cardinael et al., 2017)	0.6 - 2.6	(Cardinael et al., 2017); (López-Díaz et al., 2017)
51	Mediterranean hills, arable	silvoarable, single trees	Walnut trees on arable land	black walnut (*Juglans nigra*)	102 trees ha^{-1} (7x14m)	lines	crop rotation	timber	40-60 years	1.44 - 4.36	(Steinacker et al., 2008); (Cardinael et al., 2017)	0.86 - 2.62	(Steinacker et al., 2008); (Cardinael et al., 2017)
52	Mediterranean hills, arable	silvoarable, single trees	Fruit tree alley	olive (*Olea europaea*)	200-400 trees ha^{-1}	lines or scattered	wild asparagus	oil, forage	annual prunings, not harvested	3.14 - 6.29	(Spinelli and Picchi, 2010)	1.57-3.14	(Proietti et al., 2014)
53	Mediterranean hills, arable	silvoarable, single trees	Olive and chickpeas	olive (*Olea europaea*)	100 trees ha^{-1}	lines	chickpeas	oil, timber	annual prunings, not harvested	1.57	(Spinelli and Picchi, 2010)	0.78	(Proietti et al., 2014)

ID	Biogeographical region	AF type	Title	Tree / hedgerow species	Trees [trees ha^{-1}], hedgerow [m ha^{-1}] or wood cover [% ha^{-1}]	Planting and management system	Crop species and products	Tree products	Year of tree harvesting	Tree and root biomass, references [t ha^{-1}a^{-1}]		Carbon storage, references [t C ha^{-1}a^{-1}]	
54	Mediterranean hills, arable	silvoarable, single trees	Intercrop cork oak plantation	cork oak (*Quercus suber*)	113 trees ha^{-1}, after 20 years: 50 trees ha^{-1}	lines	crop rotation	cork, timber	80 years	1.46 - 4.29	(Palma et al. 2014)	0.34-1.29	(Palma et al. 2014)
55	Mediterranean mountains, grassland	silvopastoral, single trees	Grazed fruit plantations	olive (*Olea europaea*), almond (*Prunus dulcis*)	250 trees ha^{-1}	lines	grazing, aromatic plants	fruits, oil	annual prunings, not harvested	olive: 3.39, almond: 2.71	(Spinelli and Picchi, 2010); (Velázquez-Martí et al., 2011)	olive: 1.97, almond: 1.36	(Proietti et al., 2014); (Lopez-Bellido et al., 2016)
56	Mediterranean mountains, grassland	silvopastoral, single trees	Grazed fruit plantations	olive (*Olea europaea*), almond (*Prunus dulcis*)	250 trees ha^{-1}	lines	grazing, legume rich mix (annual self seeding species)	fruits, oil, nuts	annual prunings, not harvested	olive: 3.39, almond: 2.71	(Spinelli and Picchi, 2010); (Velázquez-Martí et al., 2011)	olive: 1.97, almond: 1.36	(Proietti et al., 2014); (Lopez-Bellido et al., 2016)
57	Mediterranean mountains, arable	silvoarable, single trees	High stem timber trees	poplar (*Populus spp*)	200 trees ha^{-1}	lines	crop rotation wheat, oilseed rape, chickpeas	timber	15 years	11.2 - 14.2	(Barrio-Anta et al., 2008), Italian, Spanish forest tables	5.76 - 7.29	(Fang et al., 2010)
58	Mediterranean mountains, arable	silvoarable, single trees	High stem timber trees	hybrid walnut (*Juglans nigra x regia*)	166 trees ha^{-1}	lines	crop rotation wheat, oilseed rape, chickpeas	timber	40 years	1.67 -6.91	(Cardinael et al., 2017); (López-Díaz et al., 2017)	1.0 - 4.15	(Cardinael et al., 2017); (López-Díaz et al., 2017)
59	Mediterranean mountains, arable	silvoarable, single trees	Intercropped fruit plantations	olive (*Olea europaea*), almond (*Prunus dulcis*), pistacchio (*Pistacia vera*)	250 trees ha^{-1}	lines	crop rotation oats, sunflower, lentils	fruits	not harvested	olive: 3.39, almond: 2.71	(Spinelli and Picchi, 2010); (Velázquez-Martí et al., 2011)	olive: 1.97, almond: 1.36	(Proietti et al., 2014); (Lopez-Bellido et al., 2016)

A4: Steppic Agroforestry practices

ID	Biogeo-graphical region	AF type	Title	Tree / hedgerow species	Trees [trees ha^{-1}], hedgerow [m ha^{-1}] or wood cover [% ha^{-1}]	Planting and management system	Crop species and products	Tree products	Year of tree harvesting	Tree and root biomass, References [t $ha^{-1}a^{-1}$]		Carbon storage, references [t C $ha^{-1}a^{-1}$]	
60	Steppic, arable	silvoarable, single trees	Mixed timber and wild fruit species	grayish oak (*Quercus pedunculiflora*), field maple (*Acer campestre*), lime (*Tilia* sp.), hawthorn (*Crataegus* sp), *Rosa* sp, blackthorn (*Prunus spinosa*)	100 trees ha^{-1}	lines	vegetable	fruits, fodder trees, timber	harvesting depends on species estimated from 25 years to 120 years.	oak: 3.11; tilia: 2.65	(Constandache et al., 2012, 2006; Costăchescu et al., 2012; Dănescu et al., 2007), Hungarian forest tables	oak: 1.59, tilia: 1.32	(Aalde et al., 2006)
61	Steppic, arable	silvoarable, single trees	High stem forest trees	poplar (*Populus* spp), willow (*Salix* spp.), black locust (*Robinia pseudoplanatus*), Pedunculate oak (*Quercus robur*), plain common and black walnut (*Juglans nigra*), common ash (*Fraxinus excelsior*), red oak (*Quercus subra*), linden (*Tilia* sp.), hazel (*Corylus avelana*), almond (*Prunus dulcis*), pine (*Pinus* sp.)	60 – 70 trees ha^{-1}	lines	vegetable	timber	70 - 90 years	poplar: 3.37-5.56, oak: 0.65-2.41; walnut: 2.21	Bulgarian, Hungarian forest tables, (Kachova et al., 2016); (Sereke et al., 2015)	poplar: 1.72 - 2.85, oak: 0.32-1.2, walnut: 1.31	(Aalde et al., 2006); (Cardinael et al., 2017)
62	Steppic, arable	silvoarable, single trees	Poplar plantation	poplar (*Populus* spp)	100 trees ha^{-1}	lines	sun-flower, cabbage, corn, pepper and eggplant, water-melon and squash, cauliflower; wheat, beans	timber	35 years	5.61 -9.28	Slovakian forest tables, (Kachova et al., 2016) ; (Barrio-Anta et al., 2008)	2.88 - 4.76	(Fang et al., 2010)
63	Steppic, arable	silvoarable, single trees	High stem forest trees	red oak (*Quercus rubra*), walnut (*Juglans regia*), alder (*Alnus* sp)	100 trees ha^{-1}	lines	corn	timber	70 - 90 years	oak:1.09-4.01; walnut: 4.19	Bulgarian, Hungarian forest tables	oak: 0.5 - 2.00, walnut: 2.51	(Aalde et al., 2006); (Cardinael et al., 2017)
64	Steppic, arable	silvoarable, single trees	Fruit orchards	walnut (*Juglans regia*), cherry (*Prunus avium*), chestnut (*Castanea sativa*)	60 – 70 trees ha^{-1}	lines	feed crops	fruits, timber	70 - 90 years	walnut: 4.19, cherry: 2.87	Hungarian forest tables, (Kachova et al., 2016) ; (Sereke et al., 2015)	walnut: 2.51, cherry: 1.55	(Cardinael et al., 2017)