Integrated assessment of silvoarable agroforestry at landscape scale

João H. N. Palma

INTEGRATED ASSESSMENT OF SILVOARABLE AGROFORESTRY AT LANDSCAPE SCALE

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In memory of Pedro Cabrita

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Preface

From 2001 to 2005 the European project Silvoarable Agroforestry for Europe (SAFE) aimed to reduce the uncertainties concerning the productivity and profitability of silvoarable systems, and to suggest European policy guidelines for agroforestry implementation. Understanding the interactions amongst the major productive components of agroforestry and its linkage to profitability at farm, landscape and regional levels were the major tasks. Although the potential environmental services of agroforestry have systematically been stipulated in the last thirty years, only few assessments exist. Evaluating the environmental services of agroforestry became one of the main goals of this PhD.

We opted for a modelling approach to achieve this goal. Finding the equilibrium between model complexity and data availability at the landscape scale in a European perspective was a major challenge.

This work cannot replace field measurements, properly designed to validate environmental effects of silvoarable agroforestry systems. The following pages, however, describe an initial flavour of what land managers can expect from these systems.

During the project, the collaboration of plant physiologists, mathematicians, foresters, agronomists and economists was the key for achieving common modelling goals. This allowed not only the up-scaling of plot scale measurements to a farm and landscape scale, but also the integration of economic results with the environmental assessment.

With this framework, an overview of profitability and environmental services of silvoarable systems allows to evaluate what can be expected from these systems as a new alternative land use in a multidisciplinary and multi-stakeholder approach. However, many other scenarios and interests can be assessed with the methods and tools here presented. The framework is flexible and could be extended to additional objectives and could be adapted to other land-use types.

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Chapter 1

General Introduction

1.1 Introduction

Agroforestry is a productive system that has been used mainly in a traditional way providing a variety of goods and services to the people of the region where it exists. Agroforestry is a collective name for land use systems and technologies where woody perennials are deliberately used on the same land management unit as agricultural crops and/or animals, either in some form of spatial arrangement or temporal sequence where there are both ecological and economic interactions between the different components (Somarriba, 1992). They can be generally divided in silvoarable (tree-crop) and silvopastoral (tree-livestock) systems being often combined in the farm strategy management.

Traditional agroforestry systems date back to roman times and beyond, they are among the earliest land-use systems developed by mankind (Lelle & Gold, 1994). The modern scientific notions of agroforestry, however, were only developed in the 1970's in the tropical regions where rapid population growth of indigenous peoples demanded that efficient productions systems be developed for both food and wood resources. As agroforestry systems were developed and refined, it also became obvious that the discipline had an important role to play in the maintenance of sustainability through its inherent resource, land and soil conservation properties (Gordon *et al.*, 1997).

In Europe, during the last three centuries, the agricultural landscape has seen a steady reduction of formerly widespread traditional agroforestry (Dupraz & Newman, 1997), despite some systems increasing in area between the two world wars (e.g. Herzog, 1998). This general reduction trend became more strongly evident after 1950, when land consolidation programs increased the size of agricultural parcels and hedges and isolated trees were removed (Eichhorn *et al.*, 2005).

As the environmental cost of intensive agriculture becomes apparent (e.g. Dealbere & Serradilla, 2004), there is an increasing interest in the promotion of ecologically sound practices (Glebe, 2003). Agroforestry favours an integrated approach that can enhance many of the biophysical cornerstones of ecologically sound agricultural production and the key is to use agroforestry systems appropriately not only for production purposes but also to achieve environmental benefits (Carruthers, 1990). Therefore, temperate agroforestry systems have become a research topic in the 1990s (Gold & Hanover, 1987). However, the number of European farmers adopting this system is still marginal.

The EU project "Silvoarable Agroforestry For Europe" investigated the European context of modern silvoarable agroforestry (SAF) between 2001 and 2005. It aimed at reducing uncertainties concerning the productivity and profitability of silvoarable systems, and to suggest European policy guidelines for agroforestry implementation (Dupraz *et al.*, 2005).

Within this broad framework, an integrated assessment of the environmental and economic performance of SAF was undertaken trough this thesis with the objective of assisting decision-makers implement ecologically sound land management practices.

The approach here embraced was built under a strong cooperation between plant physiologists, mathematicians, foresters, agronomists and economists which contributed to some of the latest scientific results on European SAF systems (Burgess et al., 2004b, Keesman et al., 2004, Graves et al., 2005a, Moreno et al., 2005, Reisner et al., 2006, van der Werf et al., 2006)

1.2 Research motivation and problem definition

European agriculture is facing the need for alternatives. The need to overcome problems of over production, low farmer's income, abandonment of rural areas and environmental pollution trough intensive production is to be dealt with a stronger relevance in the near future because recent developments of the European policy decouples direct aids from production and steer the support into stronger sustainable use of natural resources (EC, 2005b). Modern silvoarable agroforestry is a potential land use solution which fits under this venture context. The system is efficient in terms of resource use (Nair, 1993) and can be both environmentally beneficial and economically profitable. This could improve agricultural sustainability, provide opportunities to diversify farm income, provide new products to the wood industry, and create novel landscapes of high value (Dupraz & Newman, 1997).

Silvoarable agroforestry research is a challenge itself. It is neither pure agriculture nor pure forestry. Both components have to be fully understood, but also compared separately and mixed. Biophysically, the research resides in the interaction (competition/synergy) between the crop and tree components to assess productivity. Ecologically, the defy is the quantification and integration of eventual benefits or drawbacks from trees in the arable field. Economically, the challenge is to evaluate if a reduction of the crop yield is outperformed by the overall profitability, also considering subsidiary situations.

The process of "integration" poses two major problems. The first concerns the scale of evaluation. Whilst examining land-use systems at a small scale (e.g. through field experiments and subsequently, through models) can improve understanding of tree and crop interactions, the most profound effect on the environment often arises from larger scale processes (Grace *et al.*, 1997). Therefore, up-scaling scientific knowledge gathered at a plot scale to the farm and landscape scale is needed to achieve an analysis that more closely resembles reality. The second difficulty concerns integrating environmental and economic analyses of land use. Although environmental and economic analyses for agroforestry have been made through previous research, they are usually assessed separately (e.g. Thomas, 1991, Dube *et al.*, 2002, Udawatta *et al.*, 2002, Burgess *et al.*, 2004b, Nair & Graetz, 2004). Only an integrated analysis, however, where both environmental and economic effects are evaluated under identical environmental and socio-economic conditions, allows to truly recognise eventual trade-offs between the potential environmental benefits of silvoarable agroforestry and its possibly lower profitability.

In the political arena, the system has to deal with two traditional well established institutions - agriculture and forestry. Both economic sectors have their own rules, regulations and also their own understanding of good management practices. In most countries, the agricultural and forestry administration are independent of each other and are run by professionals of different educational background (i.e. agronomists and foresters). Agroforestry is therefore an opportunity to explore the multidisciplinary and open minds of decision makers and part of the motivation of this research is directed to these.

1.3 Objectives

The overall objective of the present work is to provide an integrated environmental and economic assessment of SAF at the farm/landscape scale along a European gradient.

To achieve this objective, three major tasks can be outlined:

1 – Elaborate a modelling framework methodology for the assessment of environmental performance at farm/landscape scale, through the use of existing models and through the development of new models to fit the demands of SAF system complexity.

2 – To develop a sampling strategy to allow for an unbiased selection of farms/landscapes in Europe, derive meaningful scenarios of SAF implementation and apply the methodology framework in the sampling sites.

3 – Combine the environmental results with the economic performance and conduct an integrated assessment.

1.4 Structure of the thesis

According to the research plan previously discussed, the core content of this document is sectioned in four stages as follows:

Chapter 2 explores the methodological approach to assess the environmental effects of SAF systems focusing on soil water erosion, nitrogen leaching, carbon sequestration and landscape diversity.

Chapter 3 applies the methodology developed in Chapter 2 to nineteen landscape test sites across Europe to investigate the environmental performance of SAF systems. A comparison is made with the traditional existing arable systems through the assessment of agroforestry scenarios with 50 or 113 trees ha^{-1} in 50% or 10% of the best or worst quality land in the farm/landscape.

Chapter 4 investigates the economic performance of the same scenarios. To fully explore the potential feasibility of SAF systems, the infinite net present value is evaluated under different payments levels.

Chapter 5 merges the main findings of Chapters 3 and 4 into a common framework, enabling a comprehensive understanding of the outputs of the SAF system in different situations.

Finally, Chapter 6 concludes the thesis by summarising the general conclusions and providing recommendations for further research.

Chapter 2

Methodological approach for the assessment of environmental effects of agroforestry at the landscape scale

Based on Palma J H N, Graves A R, Burgess P J, Keesman K J, van Keulen H, Mayus M, Reisner Y, Herzog F, 2006, Methodological approach for the assessment of environmental effects of agroforestry at the landscape scale, Ecological Engineering, accepted

2.1 Abstract

Silvoarable agroforestry, the deliberate combined use of trees and arable crops on the same area of land, has been proposed in order to improve the environmental performance of agricultural systems in Europe. Based on existing models and algorithms we developed a method to predict the environmental effects of SAF at a farm- and landscape-scale. The method comprised of an assessment of soil erosion, nitrogen leaching, carbon sequestration and landscape diversity and allowed the comparison of the environmental performance of SAF with arable systems using these four indicators.

The method was applied to three landscape test sites of 4 km x 4 km each in Spain, France and in the Netherlands, and compared different levels of agroforestry adoption on farmland of different potential productivity. Silvoarable agroforestry was predicted to reduce soil erosion by up to 70%, to reduce N leaching by 20-30%, to increase C sequestration over 60 years by up to 140 t C ha^{-1}, and to increase landscape diversity up to four times. The method developed was executed with widely available landscape and farm structural data and can therefore be applied to other regions in order to obtain a broader assessment of the environmental performance of silvoarable agroforestry systems.

2.2 Introduction

Silvoarable agroforestry (SAF) involves the deliberate combination of trees and arable crops on the same land management unit in some form of spatial arrangement or temporal sequence, such that there are significant ecological and economic interactions between trees and arable components (Sinclair, 1999). In temperate environments, SAF has recently attracted interest due to potential environmental benefits as compared with arable systems (Herzog, 2000), especially as reducing negative environmental impacts of agriculture has become a major concern of the European Union's Common Agricultural Policy (CAP) (Buller *et al.*, 2000, Baldock *et al.*, 2002). SAF production systems are also efficient in terms of resource use (Nair, 1993) and are therefore proposed as innovative agricultural production systems that can be both environmentally beneficial and economically profitable. This would improve agricultural sustainability, provide opportunities to diversify farm income, provide new products to the wood industry, and create novel landscapes of high value (Dupraz & Newman, 1997).

Carruthers (1990) stated that agroforestry is an integrated approach that can enhance ecologically-sound agricultural production and achieve environmental benefits. Many authors support the view that environmental value can be gained using agroforestry in a European context (e.g. Herzog, 1998, Shakesby, 2002). Their statements, however, either relate to observations made in traditional agroforestry systems or are based on conceptual considerations. No systematic investigation of the environmental performance of modern SAF has been conducted so far.

In the context of an European research project of silvoarable agroforestry (SAFE, 2001), four environmental benefits, which can be expected from SAF, were investigated:

a) ***Reduction of water-induced soil erosion*** (hereafter called soil erosion) which can preserve productive soil functions and mitigate the pollution of surface waters with soil particles and absorbed phosphorus and pesticides;

b) ***Reduction of nitrate leaching*** through the formation of a “safety net” of tree roots under the crops and increased water uptake of the system;

c) ***Carbon sequestration*** through the storage in wood not used for combustion;

d) ***Increase of landscape biodiversity*** due to an increased availability of habitats for wild species.

The majority of environmental modeling tools are developed at the point scale, where ecological processes are best understood (Visser & Palma, 2004). However, analysis at higher scale can better explain environmental phenomena (Grace *et al.*, 1997), and this is particularly the case with agroforestry due to the spatial interaction of tree and crop components. Moreover, agroforestry will typically form only one of several systems of a farm (which may also comprise grassland and arable rotation) as well as of a landscape (which consists of a mosaic of different land-use types).

Therefore, modeling approaches are required which can bridge the gap between the point and the farm- and landscape-scale. To do this, the level of detail of the models needs to be adapted to the spatial resolution of the investigation, in order to minimize modeling error. Figure 1 illustrates that increasing model complexity and spatial resolution can be associated with an increase in error due to additional data requirements. At the landscape scale, this may be because the data required to derive process-based models of high thematic and temporal resolution are not available and need to be estimated. Hence, at the landscape scale, it is more appropriate to use algorithms which integrate existing knowledge about the processes and are limited to the main governing factors.

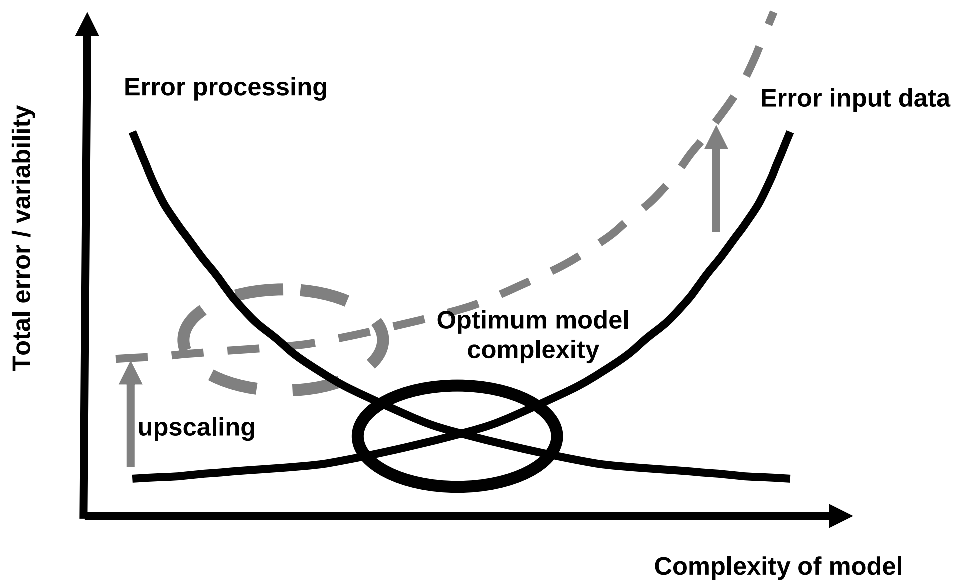


Figure 1: Relationship between model complexity and total error in the up-scaling process (Source: Wenkel & Schultz, 1999, modified).

In order to assess the previously mentioned environmental effects of SAF, we developed assessment tools based on existing models and algorithms, which were applied in landscape test sites (LTS) of 4 km x 4 km over a range of geographic situations from Mediterranean to temperate Europe. In this paper, a method to assess the selected environmental effects of SAF is explained and illustrated with results from three LTS located in Spain, France and in the Netherlands. These LTS are part of a larger sample described in Chapter 3 (*section 3.4, page 35*).

2.3 Material and methods

2.3.1 Data acquisition and processing

The investigation was conducted for LTS in Torrijos (Spain), Champlitte (France) and Scherpenzeel (The Netherlands). For each LTS, aerial photographs, taken between 1999 and 2004, were collected and the land use digitized. Soil properties were derived from existing soil maps or through field work and a digital soil map was generated for each LTS. Digital Elevation Models (DEM) were collected from national sources or developed by digitizing the contour lines of topographic maps. All spatial information was stored and processed in geographic information systems (ArcGIS – ArcInfo© and ArcInfo WorkStation© 8.3). Daily and monthly weather data (temperature, precipitation and solar radiation) were generated using Cligen 5.2 (in Lane & Nearing, 1995) from data for the nearest climate station to each LTS, compiled by Global Data Systems (GDS, 2005). Different sources of national agricultural statistics were used to complement data from the Farm Accountancy Data Network (FADN) (EC, 2003) and determine the types and typical size of farms present in the LTS.

The main climatic parameters governing resource capture, growth and production in agroforestry systems were assumed to be precipitation, solar radiation and temperature (van der Werf *et al.*, 2006). Temperature and precipitation were considered to be homogeneous within the LTS, while solar radiation was assumed to depend on the slope profiles derived from the DEM. The landscape solar radiation grid was calculated with DiGEM (Conrad, 1998) and the radiation in each grid cell was expressed as a proportion of the radiation obtained in a flat, non shaded grid cell.

The main soil property affecting tree and crop yields was assumed to be the available soil water content. This was estimated from values for soil depth and the soil water release curves identified for different soil textures (van Genuchten, 1980, Wösten *et al.*, 1999).

Table 1: Properties of landscape test sites and hypothetical farms for Torrijos in Spain, Champlitte in France and Scherpenzeel in the Netherlands.

Country	Spain		France		The Netherlands
Location	Torrijos		Champlitte		Scherpenzeel
Latitude (°)	39.89 N		47.64 N		52.57 N
Longitude (°)	4.39 W		5.58 E		6.34 E
Altitude (m)	500		300		0
Mean temperature (°C)	15.5		8.5		9
Solar radiation (MJ m^{-2})	5560		4940		3710
Rainfall (mm)	348		773		801
Farm size (ha)	63		130		10
Land unit	1	2	1	2	1
Quality	worst	best	best	worst	n.a.
Area (ha)	10	56	68	62	10
Radiation (%)	101	100	103	103	100
Soil type	medium	medium	medium	medium-fine	coarse
Soil depth	140	140	140	35	140
Tree species	Holm oak (*Quercus ilex*)	Holm oak (*Quercus ilex*)	Wild cherry (*Prunus avium*)	Walnut (*Juglans* hybr.)	Poplar (*Populus* spp.)
Crop rotation	w/f	w/w/f	w/w/w/w/w/m	w/w/o	s

Note: w, wheat; f, fallow; o, oilseed rape; m, grain maize; s, silage maize.

In order to account for spatial variability in solar radiation and available soil water content within each LTS, the LTS was divided into land units (LU) using cluster analysis (Ball & Hall, 1965, Richards, 1986) considering both, solar radiation and available soil water content as variables. Subsequently each LU was characterized by a mean proportion of total solar radiation, the major soil texture and soil depth. This resulted in two LU of different productivity for Torrijos and Champlitte, whereas Scherpenzeel resulted to be homogenous (Table 1). All the assessments (except for landscape diversity) were restricted to arable land as this was considered the target area for SAF. The land units at Torrijos and Champlitte were ranked according to potential productivity, and the crop rotation and agroforestry tree species for each LU was decided in workshops with experts and local stakeholders. The size of a typical farm within each LTS was derived from the FADN (EC, 2003) and from local statistics.

The environmental assessments were undertaken assuming a 60-year rotation of the agroforestry system. Because crop yields within an agroforestry system decline as the trees increase in size and intercept more solar radiation, it was assumed that farmers would stop arable cropping when it was unprofitable. The cut-off point was estimated from a five-year moving average of profitability (*See section 4.3.5.1*, *page 62*).

2.3.2 Assessment of soil erosion

Erosion processes and concepts are well-described (e.g. Morgan, 1995, Terrence *et al.*, 2002) and numerous soil erosion models have been developed (e.g. Wischmeier & Smith, 1978, Morgan *et al.*, 1998). Our assessment was based on the revised universal soil loss equation (RUSLE) (Renard *et al.*, 1997):

$$E = R \cdot K \cdot LS \cdot C \cdot P \qquad (\textit{Equation 1})$$

where: E is the soil loss (units: t ha^{-1} a^{-1}); R is the rainfall erosivity factor calculated over one year (units: MJ mm ha^{-1} h^{-1} a^{-1}); K is the soil erodibility factor (units: t h MJ^{-1} mm^{-1}); LS is the slope-length factor; C is the cover management factor and P is the erosion control practice factor. LS, C and P are unitless.

The R-factor was calculated according to van der Knijff *et al.* (2000), based on a fuzzy interpolation between two models (one for Northern Europe and the other for Southern Europe), which enabled the calculation of the R-factor for any latitude of Europe based on mean annual precipitation. For simplicity, precipitation was assumed to be uniform within each LTS, although this may create some error (Lima *et al.*, 2003). The K-factor was derived for each soil map unit based on the texture of the top horizon of the soil (Römkens *et al.* quoted in Renard *et al.*, 1997). The Arc Macro Language (AML) used in ArcInfo© and developed by van Remortel *et al.* (2001) was used to compute the LS-factor.

Because SAF has an arable and a forestry component, Equation 2 was developed to calculate the C-factor (C) for agroforestry:

$$C = [Cover_a \cdot C_a] + [Cover_f \cdot C_f] \qquad (\textit{Equation 2})$$

where $Cover_a$ and $Cover_f$ are the proportions of the total area occupied by the arable and forestry component respectively (0-1), and C_a and C_f are the related C-factors for the arable and forestry component. The values of $Cover_a$ and $Cover_f$ depend on the distance between the tree rows and on the tree row strip width. In the scenarios studied, it was assumed that the agroforestry system comprised 113 trees per hectare and $Cover_a$ and $Cover_f$ were assumed to be 0.91 and 0.09 respectively. The value of C_f was computed according to Dissmeyer & Foster (1980), and C_a was determined for each crop type based on Meyer (1996) and Feldwisch (1998). When the arable rotation was stopped due to unprofitability, C_a took the corresponding value for a grass cover.

2.3.3 Assessment of nitrate leaching

Although the nitrogen cycle in agricultural systems is complex (Whitehead, 1995), relatively simple equations for nitrate leaching can differentiate between different land-use systems at the regional scale. Using the approach of Feldwisch *et al*. (1998), the quantity of leached nitrogen (N_{leach}; units: kg ha^{-1} a^{-1}) was determined from:

$$N_{\text{leach}} = 4.43 \cdot N_{\text{bal}} \cdot EF \qquad \textit{(Equation 3)}$$

where: N_{bal} is the nitrogen balance (kg ha^{-1} a^{-1}), and EF is the annual soil water exchange factor (unitless).

The value of EF depends on the calculated annual flow to groundwater (F_{gw}; units: mm), and the soil water content at field capacity (FC; units: mm) (Equation 4).

$$\text{If: } \frac{F_{gw}}{FC} \geq 1\text{, then } EF = 1 \qquad \textit{(Equation 4)}$$

$$\text{If: } \frac{F_{gw}}{FC} < 1\text{, then } EF = \frac{F_{gw}}{FC}$$

Annual values for groundwater recharge were determined by summing daily values for F_{gw} derived from a process-based biophysical model called Yield-SAFE (van der Werf *et al.*, 2006) which was parameterized and calibrated for the tree species and crop rotation at each LTS (*see section 3.4.1, page 40*).

The value of the nitrogen balance (N_{bal}) was determined on an annual basis from:

$$N_{bal} = (N_{fert} + A_{dep} + N_{fix} + N_{min}) - (D + V + U + I) \qquad \textit{(Equation 5)}$$

where: N_{fert} is the addition of nitrogen fertilizer (mineral and organic); A_{dep} is the atmospheric deposition; N_{fix} is the biotic nitrogen fixation; N_{min} is the mineralization; D is the denitrification; V is the volatilization; U is the crop/tree uptake, and I is the, immobilization; all units in kg N ha^{-1}a^{-1}.

In long-term assessments with a regular cropping pattern, a steady state equilibrium is expected between mineral nitrogen released by the soil (mineralization) and the amount of nitrogen annually returned to the soil in the form of organic matter (immobilization) (Noy-Meir & Harpaz, 1977, Vlek *et al.*, 1981). Equation 5 can therefore be simplified to:

$$N_{bal} = (N_{fert} + A_{dep} + N_{fix}) - (D + V + U) \qquad \textit{(Equation 6)}$$

During the SAF rotation, tree growth and the later conversion to permanent grassland may disturb the N_{min}-I equilibrium trough the addition of organic matter (leaf fall, grassroots). However, Yield-SAFE did not allow to model these aspects. We assumed that farmers would not account for the slightly increased nitrogen availability under SAF due to leaf fall whereas under grassland, no nitrogen application was presumed.

The value of N_{fert} is usually difficult to obtain in studies with a large geographic scope. We therefore adopted the approach used by Van Keulen (1977, 1982) for determining a relationship between yield and fertilizer inputs for given soil properties. This allowed N_{fert} to be derived from the crop and tree yield values predicted by the Yield-SAFE model (*see 3.4.1, page 40*). For a given crop and tree yield, the nitrogen uptake (U; units: kg N ha^{-1}); was estimated as:

$$U = \begin{cases} \dfrac{Y_c}{\alpha} + \lambda B_t & \text{if} \quad Y_c < \dfrac{Y_{\max}}{2} \\[2ex] \dfrac{4Y_c - Y_{\max}}{2\alpha} + \lambda B_t & \text{if} \quad Y_c \geq \dfrac{Y_{\max}}{2} \end{cases} \qquad \textit{(Equation 7)}$$

where: Yc is the crop yield (unit: kg ha-1); Ymax is the maximum crop yield (unit: kg ha-1); Bt is the above-ground tree biomass (unit: kg ha-1); α is the slope from quadrant "a" in van Keulen (1982), and λ is a conversion factor to derive tree nitrogen uptake from Bt. The value of α is dependent on the biomass of the straw (S; unit: kg ha-1) and the harvested product (Yc). A content of 1% and 0.4% N in the grain and straw was assumed respectively (van Keulen & Wolf, 1986) (Equation 8):

$$\alpha = \frac{1}{0.01 + 0.004\dfrac{S}{Y_c}} \qquad \textit{(Equation 8)}$$

The value of λ is dependent on the root to shoot ratio of the tree (*RSR*; unitless), and we assumed 0.66% and 0.41% concentration of N in the tree above ground and below ground biomass respectively (Gifford, 2000a, 2000b) (Equation 9). A root to shoot ratio of 0.25 was assumed as proposed by the International Panel on Climate Change (IPCC, 1996) for broadleaved tree species.

$$\lambda = 0.0066 + (0.0041\ RSR) \qquad \textit{(Equation 9)}$$

The fertilizer application was then estimated by:

$$N_{fert} = \frac{U}{\beta} \qquad (Equation\ 10)$$

where β, the recovery factor, is a fraction between 0.5 and 0.8 depending on the management of nitrogen application (van Keulen, 1977, 1982, van Keulen & Wolf, 1986). In all LTS, β was assumed to be 0.65.

A_{dep} was obtained by summing values of oxidized and reduced nitrogen deposition from EMEP (2003). Values for denitrification (D) were derived from reference tables (Feldwisch *et al.*, 1998) and available water table information. Where no information about the water table in the LTS was available, an average value for D was adopted (30 kg N $ha^{-1}a^{-1}$). As organic fertilization was not considered separately, volatilization (V) was derived from mineral N application, as in van Keulen *et al.* (2000) and estimated as 5% of N_{fert}. As there was no legume crop modeled, N_{fix} was estimated for non symbiotic organisms as 1 kg N ha^{-1} a^{-1} (Wild, 1993).

2.3.4 Assessment of carbon sequestration

Carbon sequestration by the trees (C_{seq}; units: kg ha^{-1}) was calculated as proposed by Gifford (2000a):

$$C_{seq} = 0.5\ (B_t + RSR \cdot B_t) \qquad (Equation\ 11)$$

where, B_t is aboveground tree biomass (kg ha^{-1}), predicted by the Yield-SAFE model (*see 3.4.1, page 40*).

2.3.5 Assessment of landscape biodiversity

The introduction of SAF into a predominantly arable landscape will generally increase the diversity of habitats in that landscape. We adopted an index which relates the share of habitat that potentially adds biodiversity to the native species that persist in rural areas. We hypothesized that SAF, with a strong interaction between the permanent (tree) component and the crop component, adds a new habitat to the arable landscape matrix (Burgess *et al.*, 2003). The habitat index (I_{hab}) was defined as:

$$I_{hab} = \frac{A_{hab}}{A_{total}} \qquad \textit{(Equation 12)}$$

where A_{hab} is the area of non-arable habitats (ha) and A_{total} is the total area (ha). The value of A_{hab} was calculated as the area sum of forest, traditional orchards, riparian strips, hedges, shrub land, permanent grassland, fallow land, permanent grassland, and SAF for each LTS.

2.3.6 Scenarios

The LTS is also representative of the hypothetical farm of the dominant type in each of the three regions. We wanted to know whether – in order to generate environmental benefits – farmers should implement SAF on a small (10%) or a large part (50%) of the farm, and whether SAF should be implemented on the most productive ("best land") or least productive ("worst land") sites. These questions were formalized in four scenarios (converting 10 or 50% of the best land, or 10 or 50% of the worst land to SAF) which were compared to the present situation ("*status quo*" arable system). In the context of soil erosion, the effect of contouring practices where farming operations follow the contour lines of the terrain and where trees could be planted along contours was also examined.

For each land unit, an appropriate SAF tree species was selected according to the trees' requirements for profitable growth (Reisner *et al.*, 2006). The crop rotation in the arable system and the crop component of the silvoarable system (Table 1 page 12) followed the same *status quo* rotation, unless the crop component of the silvoarable system became unprofitable, in which case grass was assumed (*see section 4.3.5.1, page 62*). Simulations were run over a standard period of 60 years, equivalent to the length of a single life cycle of oak (*Quercus ilex*), walnut (*Juglans* hybr.) and wild cherry (*Prunus avium*) and to three growth cycles of 20-years each for poplar (*Populus* spp.).

2.3.7 Model results interpretation

The interpretation of the results is to be focused on the relative differences between scenarios rather than on the absolute values. The assessments assumed simple interpolation between plot-, farm-, and landscape- scales, and the appropriateness of this up- and down-scaling has been debated (Bierkens *et al.*, 2000, Stein *et al.*, 2001, Vachaud & Chen, 2002, Visser &

Palma, 2004). However, scale research will not be discussed here, although it is recognized as an important issue in model predictions. The objective of this paper was to develop a set of assessment tools and algorithms for major environmental indicators – not to estimate absolute values of soil loss, nitrate leaching, carbon sequestration and landscape diversity. The emphasis therefore is on possible differences among alternative land-use types, although absolute values are indicated to judge the order of magnitude of the computed values.

2.4 Results and discussion

2.4.1 Validity of the approach

The time frame of assessment was 60 years, longer than the duration of any European silvoarable agroforestry experiment. Moreover, investigations in existing experimental plots mostly deal with productivity (Burgess *et al.*, 2004b), and data on environmental performance of SAF systems are scarce. We based the validity of the modeled results on experimental evidence when possible, but we also had to rely on information from the literature.

The importance of taking the uncertainty in model predictions into account is increasingly recognized (Power, 1993, Wallach & Génard, 1998). Uncertainty analysis is an evaluation approach for measuring the reliability of model predictions in order to apply results in decision making or in land use evaluation. The analysis is performed to reduce the model output imperfections through recognition of possible model improvements. This can be achieved by identifying the essential processes of the model and by investigating which algorithms of the model may need further improvement (Wallach & Génard, 1998, Keesman & Stappers, 2004). Our investigation, however, was focused on identifying differences between scenarios rather than obtaining precise predictions. In agreement, the estimation of uncertainty in the results of the environmental assessments was rather descriptive and qualitative.

However, prior to the application of the newly developed models, the different underlying (sub) tools and algorithms have been evaluated. The evaluation consisted of a rigorous parameterization phase (implementing expert knowledge), a sensitivity analysis, calibration to many different sites and plant species and/ or a validation phase with experimental data.

2.4.2 Assessment of soil erosion

The calculated soil loss rates in the arable plots of the LTS ranged from 0.5 to 1.8 t $ha^{-1}a^{-1}$. These are of a similar magnitude to those indicated in the European soil erosion map (van der Knijff *et al.*, 2000). Although absolute values from an empirical model, that has not been locally calibrated, should be interpreted with caution (Centeri, 2003), the outputs from RUSLE can still indicate relative differences between alternative land-use types (van Remortel *et al.*, 2001).

In Torrijos and assuming no contouring, RUSLE predicted an annual soil loss of about 1.8 and 0.8 t ha^{-1} for the high (LU2) and low (LU1) quality land, respectively in the arable system (Figure 2). The fact that the predicted soil erosion was greater on high- than on low-quality land was primarily due to a more intensive rotation on high-quality land. Assuming contouring, the corresponding values were only 1 and 0.5 t ha^{-1}. The impact of SAF assuming contouring decreased these values to 0.3 and 0.1 t ha^{-1} respectively. A similar benefit has been shown for hedgerow intercropping, where soil erosion was reduced by up to 90% on gentle slopes in Nigeria, and by 45-65% on steep slopes in maize systems in Colombia (Young, 1989). The use of RUSLE did not account for gully erosion. In fact, if agroforestry is implemented without contouring, the probability of gully erosion along the tree strips could be increased due to greater erosivity of water drops under the tree canopy (Young, 1989) and this could again compensate for the reduction of soil erosion achieved through SAF.

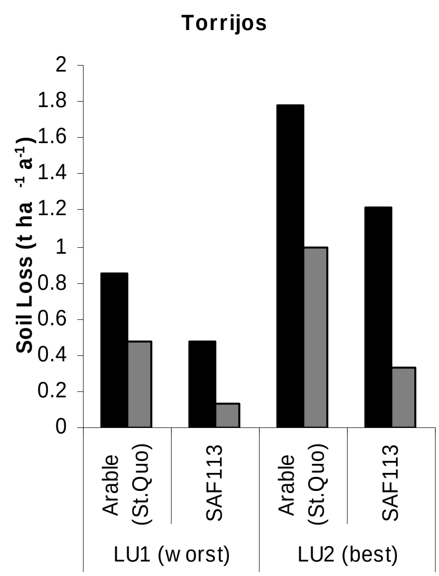


Figure 2: Estimated soil loss, at plot scale, for arable systems (St. Quo) and agroforestry (SAF_{113}) in the Torrijos landscape test site, central Spain. LU – Land Unit; See Table 1 for description of rotations.

By using the proportions of the different LU in each LTS (Table 1, page 12), the mean annual soil loss was estimated for the arable system for each LTS with and without contouring (Figure 3). Erosion rates were predicted to be similar in Champlitte and Torrijos (0.8 - 1.8 t ha^{-1}) and lower (0.3 - 0.5 t ha^{-1}) in Scherpenzeel. Contouring practices were consistently projected to reduce erosion. The greatest reduction in soil erosion (-72%) was predicted for Champlitte by combining contouring with SAF on 50% of the farm (Figure 3).

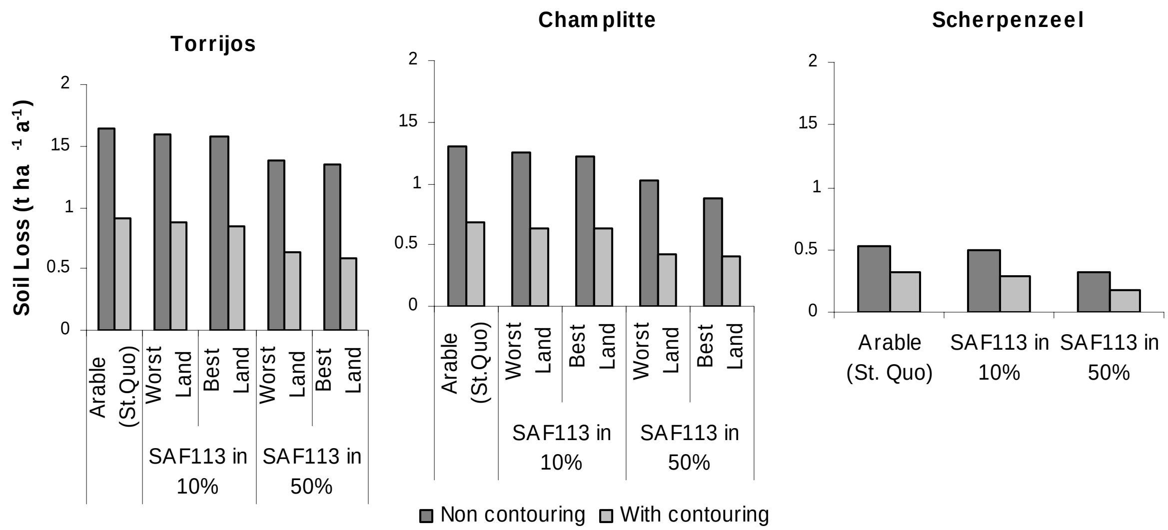


Figure 3: Estimated soil loss, at farm/landscape scale, for Torrijos (Spain), Champlitte (France) and Scherpenzeel (the Netherlands) for non-contouring and contouring practices. See Table 1 for crop rotations and tree species, section 2.3.6 for definition of scenarios.

2.4.3 Assessment of nitrogen leaching

The assessment of nitrogen leaching was based on tree and crop yields over a rotation of 60 years derived from the Yield-SAFE model (van der Werf et al., 2006) which was parameterized and calibrated for the selected tree and crop species in each LTS (Burgess *et al.*, 2005). For the low quality land unit in Champlitte, annual crop yield in the arable system ranged from 1.8 to 5.8 t ha^{-1} for wheat and 2.4 to 3.7 t ha^{-1} for oilseed (Figure 4a), and tree yield of walnut was assumed to be 69 m^3 ha^{-1} after 60 years (Figure 4b). This assumed optimum availability of nutrients.

Nitrogen input (Figure 4c) was estimated from biomass production. In the SAF system, although nitrogen uptake by the trees increased with time, this did not compensate for the reduced nitrogen uptake in the arable component and consequently total uptake in the SAF system was lower than in the arable system (Figure 4d). However, evapotranspiration for the SAF system was predicted to exceed that for the arable system, resulting in less groundwater recharge (Figure 4e) and reduced vertical transport of nitrogen. At year 40 the rotation was stopped due to economic restrictions (Figure 4a) resulting in a stop of nitrogen fertilization

(Figure 4c). As a consequence, predicted cumulated nitrogen leaching over 60 years was reduced by 40% (Figure 4f).

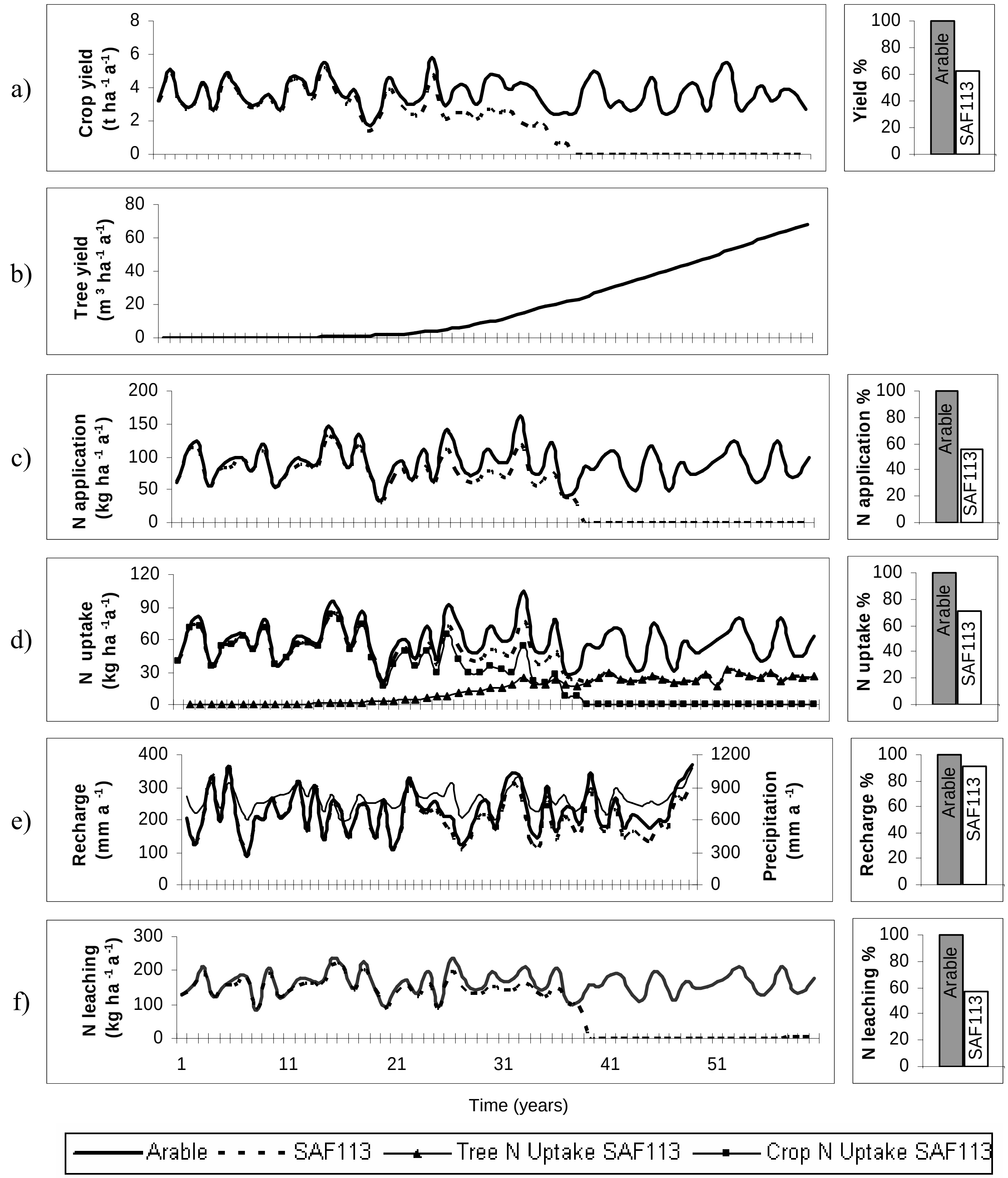


Figure 4: Comparison, at plot scale (LU2), between arable system and agroforestry (SAF_{113}) in the Champlitte LTS, east France. Tree: wild cherry; crop rotation: Wheat-Wheat-Oilseed rape. Soil texture: medium; Soil depth: 35 cm. a) Crop yield; b) Tree yield; c) N application; d) N uptake; e) Precipitation and recharge; f) N leaching. Bar graphs: Relative cumulative results for 60 years.

This approach assumed that N fertilizer application was always well-matched to the yield obtained. This assumption, which holds for both the arable and silvoarable scenarios, is probably realistic, as farmers do modify nitrogen fertilizer management in response to variations in climatic conditions and yield expectations (Kowalenko et al., 1989). The reduction in crop yield (Figure 4a) caused by increasing competition for water and light from the tree is a predictable effect that farmers can take into account when calculating fertilizer input. The calculated relative differences in N-leaching (Figure 4f) among the scenarios are therefore plausible.

Nitrogen application rates predicted for the three LTS (Figure 5) were generally lower than or similar to values in the literature. Predicted mean annual application rates for Torrijos were 40 and 36 kg in land units 1 and 2, respectively. This is within the range reported by Sadras (2002) for rainfed Mediterranean conditions. In Champlitte, the predicted mean annual applications were 153 kg in LU 1 and 90 kg in LU 2 (Figure 4c). These values are lower than a mean annual application of 160 kg from nitrogen fertilization statistics for France (Casagrande & Chapelle, 2001). In Scherpenzeel the model predicted a mean annual application of 160 kg for forage maize. Farmer interviews conducted in the same LTS indicated annual applications of 383 kg (Herzog *et al.*, 2005).

The predicted mean annual nitrogen leaching under the arable *status quo* was 0, 100 and 150 kg N ha^{-1} in Torrijos, Champlitte and Scherpenzeel respectively. No leaching was predicted at Torrijos as there was no groundwater recharge and this result agrees well with the general perception that leaching from deep soils under rainfed agriculture in the Mediterranean climate is negligible (Seligman *et al.*, 1992, Sadras, 2002). Typical values for annual N leaching from temperate European locations are 10 to 80 kg ha^{-1} (Nemeth, 1996, Hadas *et al.*, 1999, Ersahin, 2001, Hoffmann & Johnsson, 2003). Slightly higher values of up to 100 kg N ha^{-1} a^{-1} were indicated by Di & Cameron (2002) and Webster et al. (2003). Schröder (1998) reported annual nitrate leaching of 50-250 kg N ha^{-1} in forage maize systems in sandy soils in the Netherlands.

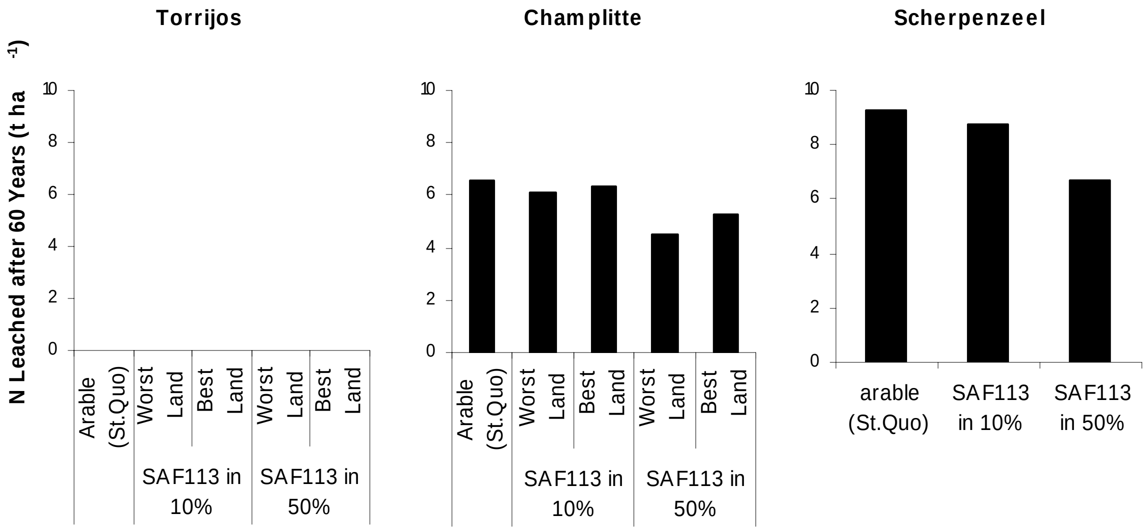


Figure 5: Estimated N leaching at the farm/landscape scale, cumulated over 60 years, for Torrijos (Spain), Champlitte (France) and Scherpenzeel (the Netherlands). Note the neglectable leaching in the mediterranean LTS due to lack of drainage. See Table 1 for crop rotations and tree species, section 2.3.6 for definition of scenarios.

The analysis predicted that implementing SAF on 50% of the farm area would reduce cumulative nitrogen leaching over a 60-year rotation by 30% at Champlitte and Scherpenzeel (Figure 5). These reductions appear less than the 40% reduction reported by Udawatta *et al.* (2002) in young temperate agroforestry systems for a three-year period. However, our approach does not account for the potential of the tree roots to recover nitrogen from below the crop rooting zone (Sanchez, 1995, van Noordwijk *et al.*, 1996, Rowe *et al.*, 2001, Udawatta *et al.*, 2002), thus leading to a conservative estimate of the potential reduction in nitrogen leaching.

The introduction of SAF was predicted to show the greatest reduction in nitrogen leaching when implemented on the highest quality land. At Champlitte, this was partly due to the predicted competitive ability of the tree species used on the best land (walnut) being higher than of the tree species on the poor land (wild cherry). For walnut, the biophysical model predicted an earlier impact on the intercrop yield than for cherry and cumulative leaching was therefore more severely reduced. However, because the worst land (shallower soil) accounted for the majority of the leaching (76%) in the whole LTS, the ponderated effect of SAF on the best land at farm/landscape scale is blurred in the cumulated results, which show the best impact in the lowest quality land (Figure 5 - Champlitte). In Scherpenzeel, where land quality was uniform, and a fast-growing tree (*Populus spp.*) was planted, leaching was reduced by 5 and 30% when SAF was implemented on 10 and 50% of the land, respectively (Figure 5 – Scherpenzeel).

2.4.4 Assessment of carbon sequestration

Generally, agroforestry systems sequester less carbon than forestry, but more than grasslands (Lasco & Pulhin, 2004). Lehmann & Gaunt (2004) and Harmand *et al.* (2004) reported that agroforestry systems are unlikely to lead to significant long-term soil carbon sequestration, as organic matter produced is relatively quickly decomposed. Therefore, the main difference in sequestration between an arable system and an agroforestry system lies in the carbon immobilized in the tree biomass (Alegre *et al.*, 2004).

Total carbon sequestered in the tree biomass for each LU was estimated using the above-ground-tree biomass predicted by the Yield-SAFE model (*see section 3.4.1, page 40)* and Equation 11 (*page 16*). Assuming an implementation of agroforestry on half of the area, over a 60-year rotation, the values of carbon were 12, 43 and 140 t ha^{-1} in Torrijos, Champlitte and Scherpenzeel respectively (Figure 6). These values are within the range of 3-60 t ha^{-1} for agroforestry systems and 190 t ha^{-1} in poplar forests reported in literature (Kürsten, 2000, van Kooten, 2000, van Kooten *et al.*, 2002, McKenney *et al.*, 2004).

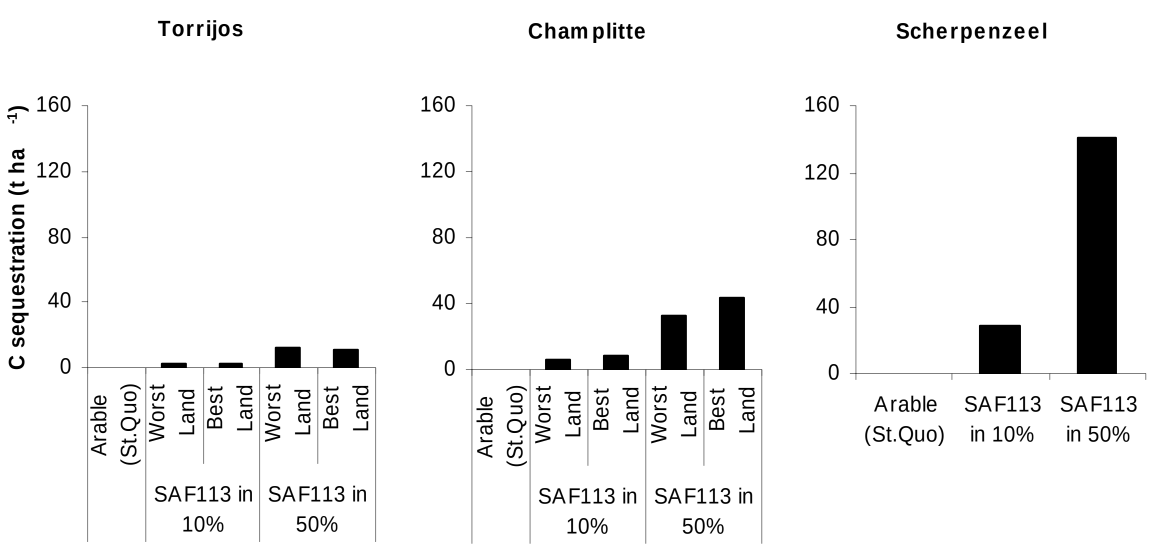


Figure 6: Estimated carbon sequestration at the farm/landscape scale, cumulated over 60 years, for Torrijos (Spain), Champlitte (France) and Scherpenzeel (the Netherlands). See Table 1 for crop rotations and tree species, section 2.3.6 for definition of scenarios.

The variation in rate of carbon sequestration among the three LTS was caused by differences in predicted growth rate of the tree species selected at each site. In the low rainfall areas of Spain, holm oak was predicted to grow slowly and sequestration was also low. At Champlitte, for walnut and wild cherry moderate levels of growth and sequestration were expected. Carbon sequestration, however, was highest for the three 20-year cycles of poplar at Scherpenzeel.

Total carbon sequestration was predicted to increase linearly with increasing proportion of land planted to agroforestry between 10 to 50% (Figure 6). Land quality had only a minor effect and further investigations are needed to substantiate these results.

2.4.5 Assessment of landscape biodiversity

Landscape diversity and species diversity are closely linked as additional land-use types, which increase the diversity of landscapes, provide habitats for additional species. Moreover, the boundaries between different land-use types (or habitats) multiply and these also consist of specific habitats for some species (Forman & Godron, 1986, Smart *et al.*, 2002).

When considering arable and SAF systems, we assumed that introducing lines of trees in homogeneous arable areas would increase the landscapes' structural diversity and thus potentially their species richness. The trees can provide habitats for some bird and arthropod species. The grassy or herbaceous strip bellow the trees consists either of sown plant species or of arable weeds; its contribution to species diversity will strongly depend on the management.

To assess the potential impact of SAF on biodiversity at landscape scale, we assumed a direct relationship between biodiversity and the proportion of the area occupied by non-arable (including SAF) and arable habitats (*see Equation 12, page 17*). This approach only accounts for landscape composition, and not for its configuration. It is therefore assumed that the increase of natural and semi-natural landscape elements will lead to an increase in biodiversity.

The relative difference between the *status quo* and the SAF scenarios depends on the habitat areas currently present. Figure 7 illustrates Equation 12 and relates the effect of converting different proportions of the arable land (10 to 90%) into SAF, and the existing proportion of non-arable habitat (5 to 90%).

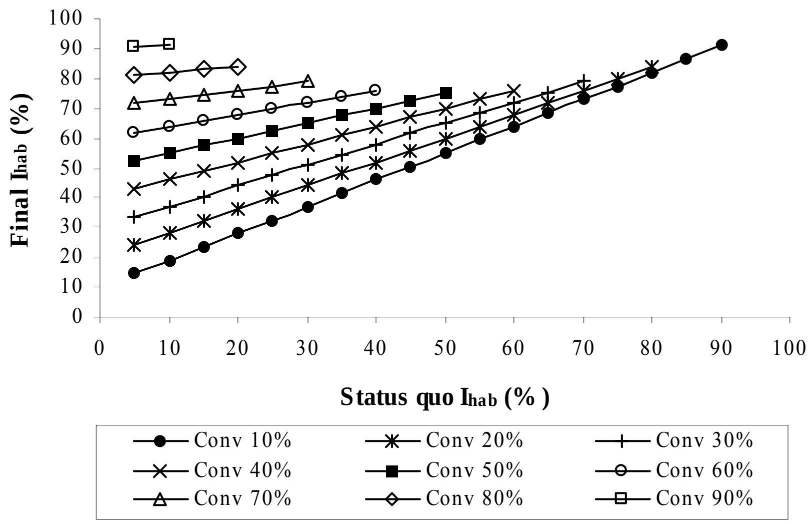


Figure 7: Relation between Status quo and Final natural and semi-natural habitat index (I_{hab}) by converting different proportions of arable land into agroforestry in the farm/landscape. See section 2.3.6 for definition of scenarios.

Consequently, in the sites under investigation, introducing SAF had the strongest impact at Scherpenzeel which had the lowest initial proportion of non-arable habitat. The conversion of 50% of the farm into SAF increased the proportion of non-arable habitat by 400% at Scherpenzeel and by 100% in Torrijos and Champlitte (Figure 8).

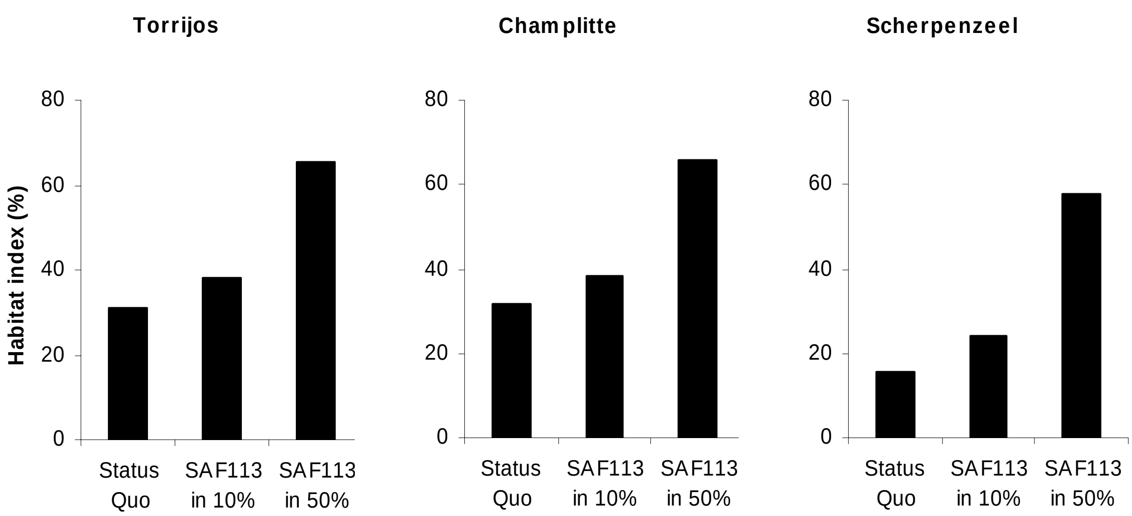


Figure 8: Estimated habitat index, at the farm/landscape scale, for Torrijos (Spain), Champlitte (France) and Scherpenzeel (the Netherlands). See section 2.3.6 for definition of scenarios.

The biodiversity of a new SAF system differs from the existing biodiversity in well established traditional agroforestry systems such as dehesas or traditional orchards (e.g. Anderson & Sinclair, 1993, Herzog, 1998, Plieninger & Wilbrand, 2001, Huang *et al.*, 2002) Their species compositions have evolved over decades, with many species depending on relatively stable conditions and being poor colonizers of new areas (Le Duc *et al.*, 1992). Nevertheless, although further research is needed, recent studies on newly established SAF systems suggest an increase in biodiversity levels (Burgess *et al.*, 2003).

2.5 Conclusions and recommendations

In Europe, positive environmental effects are expected from new land-use systems. The investigation of the environmental performance of land-use systems through experiments, however, is costly – especially at landscape scale. If trees are involved, long–term experimentation requires many years before results are available. Initiation of such experiments becomes increasingly difficult (Poulton, 1995). Therefore the modeling approach described here provided an appropriate method for assessing the environmental effects of agroforestry.

We opted for a broad view which covered four different environmental indicators (soil erosion, nitrogen leaching, carbon sequestration, landscape biodiversity), is applicable over a large geographic range (from Mediterranean to temperate Europe) and is based on the spatial and economic data that are generally available (except for the soil maps in Torrijos, which were based on field work).

Although the model results appear plausible in view of available information from literature, they can be further improved. Erosion could be assessed for different types of tree strip management and algorithms accounting for gully erosion could be added. The nitrogen leaching assessment could be improved by adding mineralization of tree litter or of pruning, which would reduce the rate of fertilization in SAF systems. Moreover, tree N uptake from below the crop root-zone would need to be accounted for. The description of the water balance could be improved by incorporating irrigation (Mayus *et al.*, 2005), this would in turn increase the scope of the model for N leaching studies. Also, in the future it should be possible to account for the potential access of tree roots to a water table; this would enlarge the range of possible situations which could be investigated. The assessment of carbon sequestration could be complemented with improved carbon allocation models. The estimation of landscape diversity could be complemented by fragmentation indices and by taking into account the spatial allocation of SAF in the landscape supported by field validation and research. More sophisticated approaches, however, require more input data of greater precision to improve the quality of the predictions. We argue that, for the purpose of a broad assessment of the effect of SAF, our approach provides a balance between modeling complexity, the number of indicators and the geographic range under investigation (Figure 1). The most important activity in improving model predictions would be local validation of input and output data.

The results suggest that SAF could reduce soil loss when introduced on high quality land, where intensive crop rotations are used. Contouring was more effective than SAF in controlling soil erosion, however, the greatest reduction in soil erosion was achieved through the combining SAF and contouring. The results also indicate that SAF could potentially reduce nitrogen leaching. Further investigations are needed to establish the order of magnitude and the influence of tree species and on productivity levels, and thus on the nitrogen cycles. Our predicted N-leaching reductions were conservative, as tree N uptake from below the crop root-zone was not considered. Whilst carbon sequestration was assumed to be zero in the arable system, some carbon is tied up in the tree component of SAF systems. Carbon sequestration was greater in fast growing species such as poplar than in the slow-growing species like walnut and wild cherry and especially holm oak, which was very slow growing. The very coarse assessment of the potential contribution of SAF to landscape diversity showed greater impact in landscapes where currently arable farming was already dominant and where only few alternative habitats existed.

To validate these preliminary conclusions and to take into account the variability of environmental and socio-economic conditions of landscapes and farms which could potentially adopt SAF systems, we will extend our approach to additional LTS in all three countries, covering thus a gradient from Mediterranean to temperate Europe. Additionally, the results will be linked to the profitability of SAF (Chaper 4) to provide an integrated environmental and economic analysis of SAF.

Chapter 3

Modelling environmental benefits of silvoarable agroforestry in Europe

Based on Palma J H N, Graves A R, Bunce R G H, Burgess P J, de Filippi R, Keesman K J, van Keulen H, Liagre F, Mayus M, Moreno G, Reisner Y, Herzog F, 2006, Modelling environmental benefits of silvoarable agroforestry in Europe, Agriculture Ecosystems & Environment, accepted

3.1 Abstract

Increased adoption of silvoarable agroforestry (SAF) systems in Europe, by integrating trees and arable crops on the same land, could offer a range of environmental benefits compared with conventional agricultural systems. Soil erosion, nitrogen leaching, carbon sequestration and landscape biodiversity were chosen as indicators to assess a stratified random sample of 19 landscape test sites in Mediterranean and Atlantic regions of Europe using computer models developed in the Silvoarable Agroforestry for Europe (SAFE) project. At each site, the effect of introducing agroforestry was examined at plot-scale by simulating the growth of one of five tree species (hybrid walnut (Juglans spp.), wild cherry (Prunus avium L.), poplar (Populus spp.), holm oak (Quercus ilex L. subsp. ilex) and stone pine (Pinus pinea L.)) at two tree densities (50 and 113 trees ha^{-1}) in combination with up to five crops; wheat (Triticum sp.), sunflower (Helianthus annuus L.), oilseed rape (Brassica napus L.), grain maize and silage maize (Zea mays L.). At landscape-scale, the effect of introducing agroforestry on 10 or 50% of the agricultural area, on either the best or worst quality land, was examined. Across the 19 landscape test sites, SAF had a positive impact on the four indicators with the strongest effects when introduced on the best quality land. The computer simulations showed that SAF could significantly reduce erosion by up to 65% when combined with contouring practices at medium (> 0.5 and < 3 t ha^{-1} a^{-1}) and high (> 3 t ha^{-1} a^{-1}) erosion sites. Nitrogen leaching could be reduced by up to 28% in areas where leaching is currently estimated high (>100 kg N ha^{-1} a^{-1}), but this was dependent on tree density. With agroforestry, predicted mean carbon sequestration through immobilization in trees, over a 60-year period, ranged from 0.1 to 3.0 t C ha-1a-1 (5 to 179 t C ha-1) depending on tree species and location. Landscape biodiversity was increased by introducing SAF by an average factor of 2.6. The implications of these changes at European scale are discussed.

3.2 Introduction

Since 1950, agricultural productivity has increased dramatically in Europe. This has been a major result of the Common Agricultural Policy (CAP) of the European Union (EU) that has successfully provided consumers with an abundant supply of a selected variety of products, whilst simultaneously the proportion of household income expended on food has declined (Grübler, 1994).

Increased agricultural output per unit area and per unit labor has been achieved using improved genetic material, increased inputs, and improved management, for example new crop varieties, the use of fertilizer and other agrochemicals, and large-scale specialized machinery. These practices were implemented together with land consolidation programs to increase the size of agricultural parcels where hedges and isolated trees were removed (Eichhorn et al., 2005). However, this has also often resulted in simplified agricultural landscapes often accompanied by loss of semi-natural and natural habitats, reduced biodiversity, soil erosion and compaction, and pollution of ground- and surface water with high levels of nitrates and pesticides (Bouma et al., 1998, Mermut & Eswaran, 2001).

Agroforestry is a form of multi-cropping which involves combining at least one woody-perennial species with a crop which results in ecological and economic interactions. Such systems are typically associated with a variety of environmental benefits and although agroforestry systems were common in Europe (Olea & Figuera, 1999, Eichhorn et al., 2005) they have been greatly reduced because of agricultural intensification (Dupraz & Newman, 1997, Herzog, 1998).

Since the 1990's, research projects have demonstrated that temperate agroforestry systems can be used with modern technology whilst preserving some of the environmental benefits associated with traditional agroforestry (Auclair & Dupraz, 1998). One form of agroforestry, here referred to as silvoarable agroforestry (SAF), is the practice of growing an arable crop between spatially-zoned trees in rows (Dupraz & Newman, 1997, Burgess et al., 2004b).

In Europe, environmental benefits are expected from new land-use systems (Baldock et al., 1993). However, investigating the environmental performance of SAF through field experiments is expensive and time-consuming because trees take decades to mature and as a consequence, the initiation of such experiments is increasingly difficult (Poulton, 1995).

Computer models provide one method for overcoming these problems. They can extrapolate research results to new combinations of biophysical and management conditions that are too complex to be studied in field experiments (Mobbs et al., 2001). However, the majority of research on temperate agroforestry systems has been undertaken to evaluate their productivity, whereas information on their environmental performance remains scarce.

To tackle this, a modeling approach was developed by Palma *et al*. (2006) to assess the environmental performance of SAF systems. This included examining the impact of SAF on soil erosion by water (hereafter called erosion), nitrogen leaching, carbon sequestration and landscape biodiversity, and uses tree and crop yields derived from a biophysical model called YieldSAFE (van der Werf et al., 2006).

The objective of this paper is to use the approach of Palma *et al*. (2006) to assess the potential environmental performance of SAF in representative climatic conditions of southern Europe (Mediterranean Spain), western Europe (France), and northern Europe (the Netherlands).

3.3 Material and methods

Randomly selected landscape test sites (LTS) in Spain, France and the Netherlands were used to model tree and crop yields on hypothetical farms for SAF at two densities (50 and 113 trees ha^{-1}, 40 x 5m and 22 x 4m respectively) on 10 and 50% of the total agricultural area, starting with either the best and worst quality land. Current agricultural land use was also modeled to provide a comparison with the status quo. YieldSAFE (van der Werf et al., 2006) was used to generate crop yields for typical crop rotations at each LTS over a 60-year time horizon. The same crop rotations were then used in SAF systems that included holm oak (*Quercus ilex* subsp. *ilex* L.) and stone pine (*Pinus pinea* L.) in Spain, and hybrid walnut (*Juglans* sp), wild cherry (*Prunus avium* L.) and poplar (*Populus* spp) in France and the Netherlands. An initial stage of the investigation involved characterizing the LTS to provide inputs for the YieldSAFE model and environmental assessment and then generating tree and crop yields for the scenarios described above.

3.4 Data acquisition and processing

Based on an environmental classification of Europe, which resulted from a statistical analysis of climatic and topographic data (Metzger et al., 2005), 21 LTS of 4 km x 4 km each were randomly selected in the dominant environmental classes of Spain (9), France (9) and The Netherlands (3). The selection was random, but was restricted to agricultural areas according to the PELCOM land cover classification (Mücher, 2000). Two LTS in France were later discarded due to lack of associated data, bringing the total to 19 LTS. In Spain the sites ranged from Alcala la Real in Andalucia in the south to St Maria del Paramo in Castilla y Leon in the north. In France, the sites ran across central France from Champdeniers in Poitou Charentes in the west to Champlitte in Franche Comté in the east. In the Netherlands the sites were located in the central (Gelderland) and eastern (Overijssel) parts of the country (Figure 9).

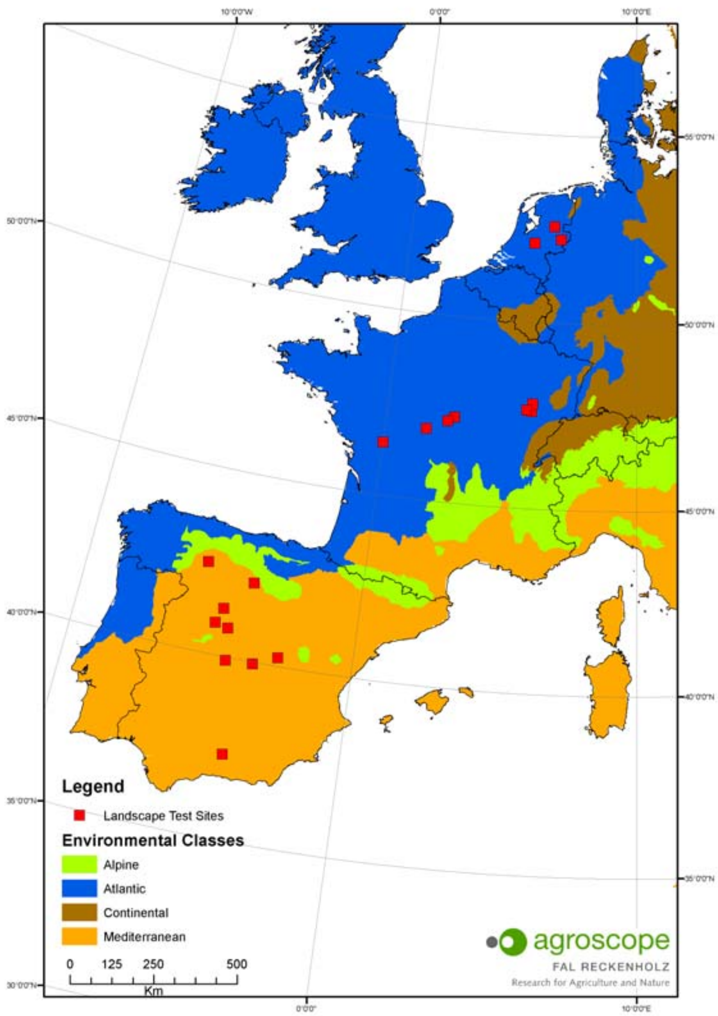


Figure 9: Landscape test sites selected covering wide biophysical characteristics based on the European environmental classification (Metzger et al., 2005). See Table 2 for the site codes.

In Spain, aerial ortho-images were obtained from the SIG Oleícola Español (MAPYA, 1999) and digital land-use data were obtained from the REDPARES project (Bolaños et al., 2003). During field surveys, land-use was updated and soil samples were taken to produce soil maps in combination with topographic details. Digital elevation models (DEM) were developed by digitizing the contour lines of topographic maps. In France, aerial photographs and DEM were acquired from IGN©, and the land-use digitized. Digital soil maps were acquired from various regional institutions. In the Netherlands, aerial photographs and land-use data were obtained from the EU GREENVEINS project consortium (Bailey et al., 2005). Digital elevation models were acquired from DLG© and digital soil maps from GeoDesk©. At each LTS, daily and monthly weather data (temperature, precipitation and solar radiation) were generated using

Cligen 5.2 (Lane & Nearing, 1995) based on reference data from the weather station nearest to the LTS (GDS, 2005). All spatial information was stored and processed in geographic information systems (ArcGIS – ArcInfo© and ArcInfo WorkStation© 8.3).

Data on temperature, radiation, precipitation and soil water availability are required to generate tree and crop yields in YieldSAFE. Precipitation and temperature were considered to be homogenous within each LTS, while solar radiation was considered to vary depending on the direction and angle of the slopes described by the DEM. A solar radiation grid was calculated for one year and each LTS with DiGEM (Conrad, 1998) and transformed into a percentage by dividing the radiation in each grid cell by the radiation obtained in a flat, un-shaded grid cell. From the soil information, available water content was estimated based on soil depth and texture to which were associated “van Genuchten” parameters assessed by Wösten et al. (1999) and volumetric water content calculated with the van Genuchten equation (1980).

To account for spatial variability in solar radiation and available soil water content within each LTS, both maps were processed using the isocluster analysis function in ArcInfo© 8.3 (Ball & Hall, 1965, Richards, 1986) resulting in up to four land units (LU) or clusters. Each LU was then characterized by its mean radiation, and its major soil texture and soil depth (Figure 10). The cluster analysis resulted in 42 land units for the 19 LTS, each potentially capable of producing different tree and crop yields (Table 2, page 39). All analyses, except that for landscape biodiversity, were restricted to agricultural land within each LU, since this land was considered to be the target area for SAF. Within each LTS, LU’s were ranked according to their potential productivity. When more than 2 LU’s were present, an intermediate quality was also given (medium). Crop rotations and agroforestry tree species were determined for each LU in workshops with experts and local stakeholders (Table 2).

Hypothetical farms were also devised for each LTS, using farm structure data from FADN (EC, 2003) and local statistics to define the total size of the farm. The area of each LU within each hypothetical farm was derived from the proportion of each LU within each LTS (Figure 10).

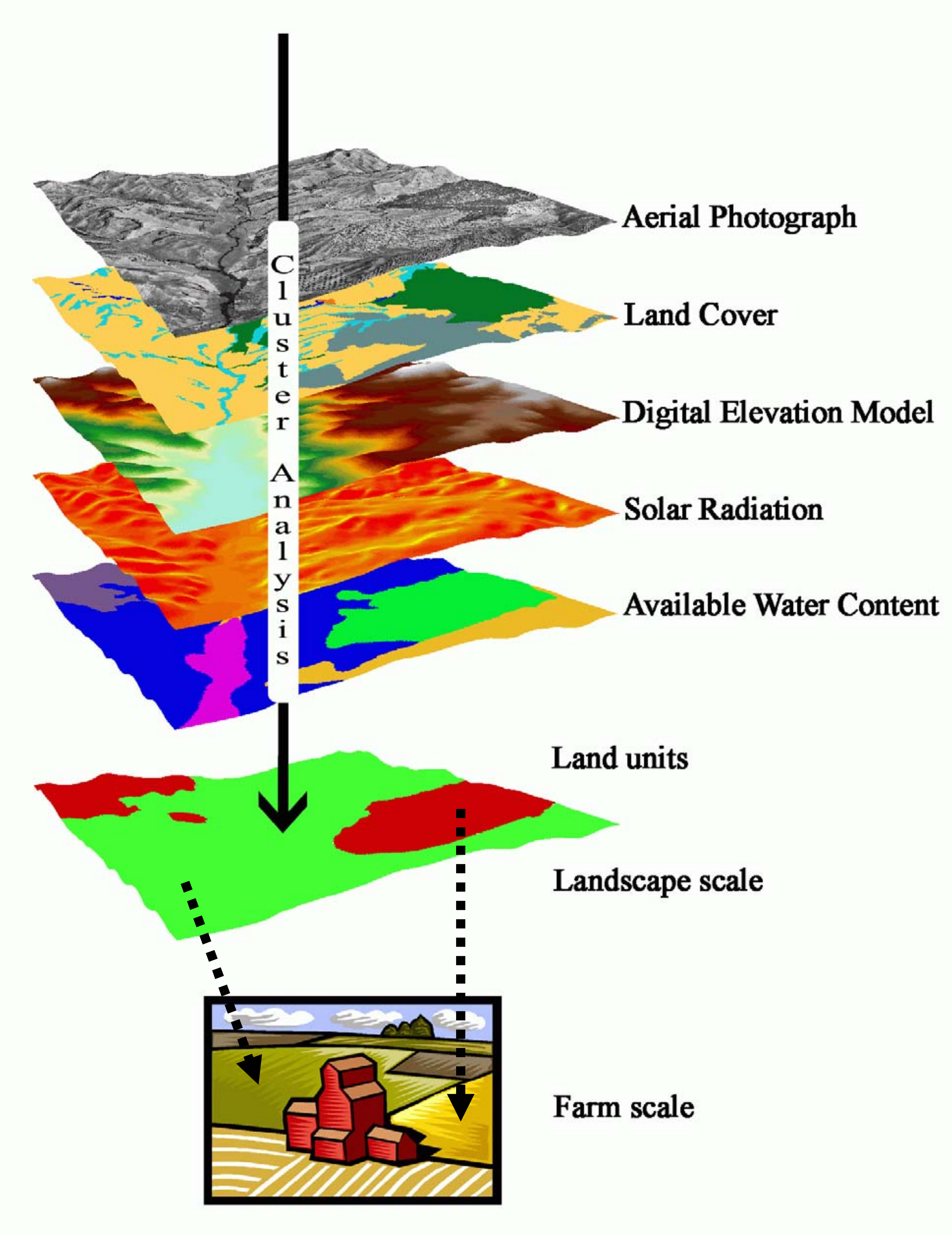


Figure 10: GIS landscape data processing for each landscape test site (LTS) to create homogeneous land units (LU), corresponding to different qualities of agricultural land of a farm (Torrijos LTS example).

Table 2: Biophysical and management characteristics of the landscape test sites in Spain, France and the Netherlands and corresponding land units (LU).

Site	Site Code	Altitude (m)	Mean Temp (°)	Area of farm (ha)	Rainfall (mm)	LU – quality	Area of LU (ha)	Radiation (%)	Soil texture (FAO)	Soil depth (cm)	Tree	Crop rotation
Spain												
Alcala la Real	ALC	1000	15	73	355	LU1-B	58	97	M	140	Oak	w/w/f
						LU2-W	15	86	M	50	Oak	w/w/f
Torrijos	TOR	500	15	63	348	LU1-W	10	101	M	140	Oak	w/f
						LU2-B	56	100	M	140	Oak	w/w/f
Ocaña	OCA	700	15	66	316	LU1-na	66	100	M	140	Oak	w/w/f
Almonacid de Zorita	ALM	900	13	66	404	LU1-B	59	97	M	140	Oak	w/f
						LU2-W	7	83	F	140	Oak	s/s/s/s/s/w/f
Cardenosa El Espinar	CAR	1000	12	58	404	LU1-W	23	93	M	140	Oak	w/w/w/f
						LU2-B	35	101	F	140	Oak	w/w/w/f
Fontiveros	FON	900	12	58	393	LU1-B	49	99	C	140	Oak	w/w/w/w/f
						LU2-W	9	98	C	140	Pine	w/w/w/w/f
Olmedo	OLM	750	12	57	410	LU1-M	5	100	C	140	Pine	w/s/f
						LU2-B	34	100	M	140	Oak	w/s/f
						LU3-W	18	99	C	140	Oak	w/s/f
St Maria del Campo	CAM	800	10	58	530	LU1-W	44	99	C	140	Pine	w/w/w/f
						LU2-B	14	99	M	140	Oak	w/w/w/w/w/f
St Maria del Paramo	PAR	800	10	59	519	LU1-B	4	100	M	140	Oak	w/w/w/s/f
						LU2-M	34	100	M	140	Oak	w/w/w/s/f
						LU3-W	21	101	M	140	Oak	w/w/w/s/f
France												
Champdeniers	CHD	200	11	94	648	LU1-B	67	100	F	80	Cherry	w/w/s/w/o/s
						LU2-W	27	100	M	120	Walnut	w/w/s/w/o/s
Chateauroux	CHT	150	11	152	587	LU1-M	32	102	F	80	Walnut	w/w/o/w/o/s
						LU2-B	23	102	F	40	Cherry	w/w/o/w/o/s
						LU3-M	86	102	M	120	Walnut	w/w/o
						LU4-W	11	100	F	40	Cherry	w/w/o/w/o/s
Fussy	FUS	200	10	80	626	LU1-W	10	101	F	40	Cherry	w/o
						LU2-B	43	103	M	80	Poplar	w/w/o
						LU3-M	27	102	F	120	Cherry	w/o
Sancerre	SAN	400	11	98	724	LU1-M	37	103	F	40	Cherry	o/w/s/w/w/w/o
						LU2-W	10	102	VF	140	Poplar	o/w/s/w/w/w/o
						LU3-B	44	101	VF	120	Cherry	o/w/s/w/w/w/o
						LU4-B	7	100	C	80	Cherry	o/w/s/w
Champlitte	CMP	300	8	130	773	LU1-W	68	103	M	140	Cherry	w/w/o
						LU2-B	62	103	MF	35	Walnut	w/w/w/w/w/gm
Dampierre	DAM	300	10	130	1072	LU1-M	64	98	M	140	Cherry	w/w/gm
						LU2-W	43	97	F	35	Cherry	w/w/w/gm
						LU3-B	23	95	MF	60	Poplar	w/gm
Vitrey	VIT	400	9	120	1084	LU1-W	46	103	M	60	Cherry	w/w/o
						LU2-B	74	103	MF	60	Poplar	w/w/gm
The Netherlands												
Balkbrugg	BAL	0	9	40	818	LU1-na	40	100	C	140	Poplar	sm
Bentelo	BEN	0	9	40	729	LU1-na	40	100	C	140	Walnut	w/w/sm
Scherpenzeel	SCH	0	9	10	801	LU1-na	10	100	C	140	Poplar	sm

Land Units (LU): B, best; M, medium; W, worst; na, not applicable. Soil type: C, coarse; M, medium; MF, medium-fine; F, fine; VF, very-fine. Crops: w, wheat; f, fallow; o, oilseed rape; s, sunflower; gm, grain maize; sm, silage maize

3.4.1 Biophysical modelling

As this chapter was co-ordinated with Chapter 4, details of the parameterization and calibration of YieldSAFE can be found in *section 4.3.3, page 59* and the biophysical production data in *sections 4.4.1 and 4.4.2, pages 66* and *68* respectively.

3.4.2 Data analysis

The environmental assessment was performed in each LU assuming a 60-year rotation for the agroforestry system. For poplar, which was assumed to have a rotation of 20 years, three successive tree crops were included. Because crop yields within an agroforestry system decline over time as the trees increase in size and compete with the crops, it was assumed that farmers would stop arable cropping when it became unprofitable. The cut-off point was estimated from a five-year moving average of profitability as described in Chapter 4 (*section 4.3.5.1, page 62)*.

For each scenario, soil erosion, nitrogen leaching, carbon sequestration and landscape biodiversity were examined using the method described in Chapter 2. Erosion was modeled with the revised universal soil loss equation (RUSLE, Renard et al., 1997), where SAF was considered to mimic strip cropping which could be implemented with or without an erosion control measure, in this case contouring. Nitrogen leaching was modeled using an equation proposed by Feldwisch et al. (1998), which uses an annual water exchange factor in the soil and the excess nitrogen potentially available for leaching. Annual excess nitrogen was estimated from tree and crop productivity, assuming optimized nitrogen fertilization which considered nitrogen contents of crop-tree biomass, soil and the nitrogen recovery capacity by crops (van Keulen, 1982). Crop and tree yields were computed in Chapter 4 and van der Werf et al. (2006) for each land unit using YieldSAFE, which also calculated groundwater recharge required to compute annual nitrogen leaching. Carbon sequestration was calculated for SAF systems only, based on the Intergovernmental Panel on Climate Change (IPCC, 1996) and Gifford relationships (2000a, 2000b) for tree biomass predicted by the YieldSAFE model. A broad evaluation of the effects of SAF implementation on landscape biodiversity was estimated using the share of habitats available to wildlife in an agricultural landscape, using classified distinctions between habitat and non-habitat in farmland.

Each environmental assessment for each LU in each LTS was then used to calculate a weighted mean at the farm scale, based on the proportion of land occupied by each LU within each hypothetical farm. These were then aggregated to provide an overall assessment of environmental effect for each scenario at each LTS.

Modelled results are representations of reality that can be statistically compared (Kleijnen, 1987). LU and LTS scale results were compared with general linear models (GLM) in STATISTICA©. Multiple comparisons between scenarios were tested with Tukey HSD (Honest Significant Difference).

3.5 Results and discussion

The result of the environmental assessment for each scenario at each LTS is shown in Table 3. Although the results are presented as mean annual values to facilitate interpretation, the annual rates are not constant over the 60 years time horizon because of variation in weather, crop rotations, and the growth of trees in the SAF systems. For example, in SAF systems, soil erosion was greatly reduced when a grass fallow was introduced at the termination of profitable cropping, since such cover is an effective means of preventing soil loss (Morgan, 1995, Reisner & Freyer, 2005). Similarly, nitrogen leaching is reduced (Whitehead, 1995). An example of the annual variability in nitrogen leaching is shown for a walnut SAF system for the LTS at Champdeniers in France (Figure 11).

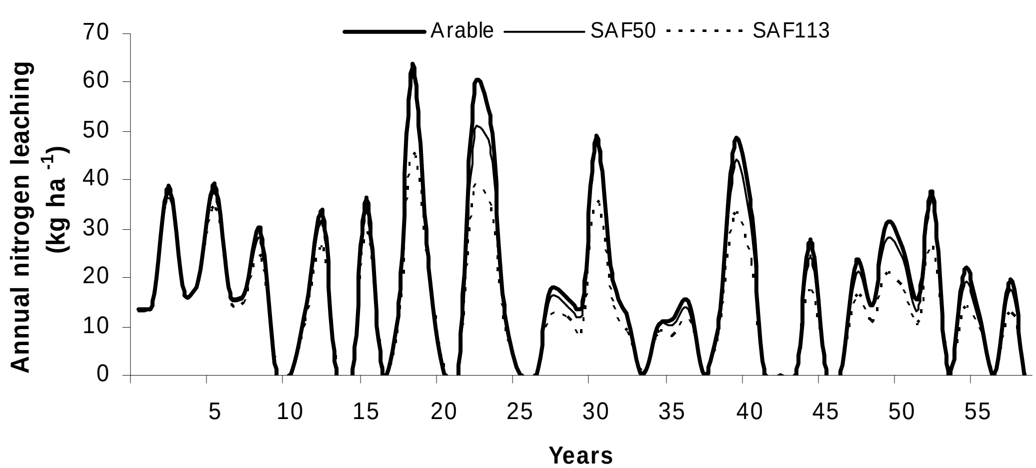


Figure 11: Predicted annual nitrate leaching at land unit scale (LU3) in the Chateauroux landscape test site over a 60-year period with walnut and a wheat-wheat-oilseed rotation. Arable reference scenario (Arable, average annual leaching 17 kg N ha^{-1} a^{-1}), silvoarable scenario with 50 trees per hectare (SAF50, 16 kg N ha^{-1} a^{-1}), silvoarable scenario with 113 trees per hectare (SAF113, 12 kg N ha^{-1} a^{-1}).

Table 3: Farm/landscape scale scenario assessment results for erosion and nitrate leaching.

Indicator	Scenarios: Erosion control practices	Scenarios: SAF Density trees ha^{-1}	Scenarios: SAF Area (%)	Scenarios: Land Quality	LANDSCAPE TEST SITES: SPAIN									FRANCE							NETHERLANDS		
					ALC	TOR	OCA	ALM	CAR	FON	OLM	CAM	PAR	CHD	CHT	FUS	SAN	DAM	CMP	VIT	SCH	BEN	BAL
Erosion (t ha^{-1} a^{-1})	Without contouring practices	Arable (Status quo)			6.0	1.6	0.0	3.4	3.4	1.2	0.0	0.7	0.3	0.4	0.4	1.1	2.1	2.9	1.3	9.7	0.5	0.5	0.3
Erosion (t ha^{-1} a^{-1})	Without contouring practices	SAF 50	10	Worst	5.9	1.6	0.0	3.3	3.4	1.2	0.0	0.6	0.3	0.4	0.4	1.1	2.1	2.8	1.3	9.7	0.5	0.4	0.2
Erosion (t ha^{-1} a^{-1})	Without contouring practices	SAF 50	10	Best	6.0	1.6	0.0	3.3	3.4	1.2	0.0	0.7	0.3	0.4	0.4	1.0	2.0	2.5	1.2	8.8	0.5	0.4	0.2
Erosion (t ha^{-1} a^{-1})	Without contouring practices	SAF 50	50	Worst	5.7	1.4	0.0	3.1	3.3	1.2	0.0	0.6	0.3	0.4	0.4	1.1	2.0	2.4	1.2	8.5	0.3	0.3	0.2
Erosion (t ha^{-1} a^{-1})	Without contouring practices	SAF 50	50	Best	5.9	1.4	0.0	3.1	3.4	1.1	0.0	0.6	0.3	0.4	0.4	0.7	1.5	2.2	0.9	5.0	0.3	0.3	0.2
Erosion (t ha^{-1} a^{-1})	Without contouring practices	SAF 113	10	Worst	5.9	1.6	0.0	3.3	3.4	1.2	0.0	0.6	0.3	0.4	0.4	1.1	2.1	2.8	1.2	9.6	0.5	0.4	0.2
Erosion (t ha^{-1} a^{-1})	Without contouring practices	SAF 113	10	Best	5.9	1.6	0.0	3.3	3.4	1.2	0.0	0.7	0.3	0.4	0.4	1.0	2.0	2.5	1.2	8.7	0.5	0.4	0.2
Erosion (t ha^{-1} a^{-1})	Without contouring practices	SAF 113	50	Worst	5.5	1.4	0.0	2.9	3.3	1.1	0.0	0.5	0.2	0.3	0.4	1.1	2.0	2.4	1.0	8.0	0.3	0.3	0.1
Erosion (t ha^{-1} a^{-1})	Without contouring practices	SAF 113	50	Best	5.7	1.4	0.0	2.9	3.3	1.1	0.0	0.5	0.3	0.4	0.4	0.7	1.5	2.0	0.9	4.8	0.3	0.3	0.1
Erosion (t ha^{-1} a^{-1})	With contouring practices	Arable			4.4	0.9	0.0	2.3	2.2	0.7	0.0	0.4	0.1	0.2	0.2	0.6	1.1	1.6	0.7	5.3	0.3	0.3	0.2
Erosion (t ha^{-1} a^{-1})	With contouring practices	SAF 50	10	Worst	3.9	0.9	0.0	2.2	2.0	0.6	0.0	0.4	0.1	0.2	0.2	0.5	1.1	1.4	0.6	5.1	0.3	0.2	0.1
Erosion (t ha^{-1} a^{-1})	With contouring practices	SAF 50	10	Best	4.2	0.8	0.0	1.8	2.1	0.6	0.0	0.4	0.1	0.2	0.2	0.5	1.0	1.3	0.6	4.7	0.3	0.2	0.1
Erosion (t ha^{-1} a^{-1})	With contouring practices	SAF 50	50	Worst	2.1	0.6	0.0	1.8	1.5	0.5	0.0	0.3	0.1	0.2	0.2	0.4	0.8	1.0	0.5	4.0	0.2	0.2	0.1
Erosion (t ha^{-1} a^{-1})	With contouring practices	SAF 50	50	Best	3.3	0.6	0.0	1.8	1.7	0.5	0.0	0.3	0.1	0.2	0.2	0.3	0.7	1.0	0.4	2.4	0.2	0.2	0.1
Erosion (t ha^{-1} a^{-1})	With contouring practices	SAF 113	10	Worst	3.9	0.9	0.0	2.2	2.0	0.6	0.0	0.4	0.1	0.2	0.2	0.5	1.1	1.4	0.6	5.1	0.3	0.2	0.1
Erosion (t ha^{-1} a^{-1})	With contouring practices	SAF 113	10	Best	4.2	0.8	0.0	1.8	2.1	0.6	0.0	0.4	0.1	0.2	0.2	1.0	1.0	2.5	0.6	4.7	0.3	0.2	0.1
Erosion (t ha^{-1} a^{-1})	With contouring practices	SAF 113	50	Worst	2.0	0.6	0.0	1.7	1.4	0.5	0.0	0.2	0.1	0.1	0.2	0.4	0.8	1.0	0.4	3.9	0.2	0.2	0.1
Erosion (t ha^{-1} a^{-1})	With contouring practices	SAF 113	50	Best	3.2	0.6	0.0	1.7	1.6	0.5	0.0	0.2	0.1	0.2	0.2	0.3	0.7	1.0	0.4	2.3	0.2	0.2	0.1
N Leaching (kg ha^{-1} a^{-1})		Arable (Status quo)			4	0	0	0	0	0	1	7	0	37	70	48	59	137	109	134	155	124	149
N Leaching (kg ha^{-1} a^{-1})		SAF 50	10	Worst	0	0	0	0	0	0	1	7	0	37	71	49	61	132	107	136	151	103	131
N Leaching (kg ha^{-1} a^{-1})		SAF 50	10	Best	0	0	0	0	0	0	1	7	0	37	70	51	59	137	106	133	151	103	131
N Leaching (kg ha^{-1} a^{-1})		SAF 50	50	Worst	0	0	0	0	0	0	1	6	0	37	76	49	68	119	97	140	135	81	113
N Leaching (kg ha^{-1} a^{-1})		SAF 50	50	Best	0	0	0	0	0	0	1	6	0	36	74	60	61	136	91	130	135	81	113
N Leaching (kg ha^{-1} a^{-1})		SAF 113	10	Worst	0	0	0	0	0	0	1	6	0	36	70	47	60	131	102	132	146	99	125
N Leaching (kg ha^{-1} a^{-1})		SAF 113	10	Best	0	0	0	0	0	0	1	7	0	36	70	49	59	126	105	127	146	99	125
N Leaching (kg ha^{-1} a^{-1})		SAF 113	50	Worst	0	0	0	0	0	0	1	5	0	33	70	45	60	113	75	118	112	74	100
N Leaching (kg ha^{-1} a^{-1})		SAF 113	50	Best	0	0	0	0	0	0	1	6	0	33	69	50	59	108	88	99	112	74	100

Table 3 (continued)

Indicator	Scenarios: Erosion control practices	SAF density trees ha^{-1}	Area (%)	Land quality	LANDSCAPE TEST SITES: SPAIN ALC	TOR	OCA	ALM	CAR	FON	OLM	CAM	PAR	FRANCE CHD	CHT	FUS	SAN	DAM	CMP	VIT	NETHERLANDS SCH	BEN	BAL
Carbon sequestration (t C ha^{-1})		Arable (Status quo)			0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	SAF 50		10	Worst	0	1	3	2	1	2	2	2	2	5	4	4	4	3	5	5	22	11	69
				Best	1	1	3	2	2	1	3	3	2	5	5	20	29	14	6	26	22	11	69
			50	Worst	3	6	14	12	7	8	11	10	9	26	22	31	22	19	25	48	108	22	139
				Best	3	6	14	12	9	7	13	14	9	26	23	100	135	41	28	128	108	22	139
	SAF 113		10	Worst	0	3	5	5	3	3	4	4	4	8	6	5	6.1	5	7	7	28	17	84
				Best	1	2	5	3	4	3	5	6	3	7	8	25	39	16	9	31	28	17	84
			50	Worst	7	12	25	23	15	16	20	21	17	38	29	39	29	29	32	63	141	34	168
				Best	5	11	25	23	18	15	23	29	17	35	32	126	179	54	44	155	141	34	168
Habitat index [%]		Arable (Status quo)			81	31	8	61	75	46	50	11	18	2	4	11	7	3	32	11	16	16	1
	SAF50 or SAF113		10	na	83	38	17	65	77	52	55	20	26	12	14	20	16	12	38	20	24	25	11
			50	na	90	66	54	80	87	73	75	55	59	51	52	56	53	51	66	56	58	58	51

***Notes:* na, not applicable. See Table 2 for landscape test sites codes.**

In interpreting the results, we focused on the relative differences between scenarios, rather than on the absolute values. However, absolute values have been tabulated to indicate the magnitude of the computed values.

3.5.1 Erosion

Predicted erosion rates at the 19 LTS for the status quo arable systems ranged from 0 to 9.7 t ha^{-1} a^{-1} (Table 3). These are of a similar magnitude than those indicated in the European soil erosion map for individual LTS locations (van der Knijff et al., 2000). Although absolute values from an empirical model that has not been locally calibrated should be interpreted with caution (Centeri, 2003), the outputs from RUSLE can still be used to indicate relative differences in soil erosion between alternative land-use types (van Remortel et al., 2001).

Introduction of SAF reduced erosion at all LTS in comparison with the arable status quo, especially when contouring was practiced (Table 3). To test significance, we grouped the LU and LTS into categories of low (< 0.5 t ha^{-1} a^{-1}), medium (> 0.5 and < 3 t ha^{-1} a^{-1}) and high (> 3 t ha^{-1} a^{-1}) erosion sites (Table 4).

Contouring is an important erosion control measure. However, the implementation of contouring alone did not significantly reduce erosion, nor did the implementation of SAF alone. However, when both measures were combined, statistical analysis suggests significant reductions in erosion at medium (> 0.5 and < 3 t ha^{-1} a^{-1}) and high (> 3 t ha^{-1} a^{-1}) erosion sites. Results at the LU scale suggest that on medium erosion sites, combining SAF and contouring could significantly reduce erosion by up to 80% from 1.6 to 0.3 t ha^{-1} a^{-1} for both tree densities and land types studied (Table 4a). Approximately the same magnitude of reduction was calculated for the high erosion sites for both tree densities, but the effect was only significant ($p<0.05$) on the best land.

Table 4: Effect on average soil loss (t ha^{-1} a^{-1}) of non-contour and contouring practices with arable cropping, and silvoarable agroforestry with 50 trees ha^{-1} (SAF50) and 113 trees ha^{-1} (SAF113), for low , medium, and high erosion sites on: a) the best and worst quality land at plot (land unit) scale and on b) 50% of the worst or best quality land at a farm (landscape test site) scale.

a) Land unit scale		Low (<0.5 t ha^{-1})		Medium		High (> 3 t ha^{-1})	
		Worst	Best	Worst	Best	Worst	Best
Non- contouring	Arable	0.4	0.3	1.5^{b}	1.6^{b}	5.8^{ab}	7.0^{b}
	SAF50	0.3	0.3	1.2^{ab}	1.3^{ab}	5.2^{ab}	4.2^{ab}
	SAF113	0.3	0.3	1.1^{ab}	1.1^{ab}	4.7^{ab}	3.8^{ab}
Contouring	Arable	0.1	0.2	0.9^{ab}	0.9^{ab}	3.8^{ab}	4.5^{ab}
	SAF50	0.1	0.1	0.3^{a}	0.3^{a}	1.4^{a}	1.1^{a}
	SAF113	0.1	0.1	0.3^{a}	0.3^{a}	1.3^{a}	1.0^{a}
	n	4	5	7	5	4	5
Stat. sig.		*NS*		*		***	

b) Landscape test site scale		Low (<0.5 t ha^{-1})		Medium		High (> 3 t ha^{-1})	
		Worst	Best	Worst	Best	Worst	Best
Non- contouring	Arable	0.3		1.7^{c}		5.6	
	SAF50 10%	0.3	0.3	1.7^{bc}	1.6^{abc}	5.6	5.4
	SAF50 50%	0.3	0.3	1.6^{abc}	1.3^{abc}	5.2	4.3
	SAF113 10%	0.3	0.3	1.7^{abc}	1.6^{abc}	5.5	5.3
	SAF113 50%	0.2	0.3	1.5^{abc}	1.3^{abc}	4.9	4.2
Contouring	Arable	0.2		0.9^{abc}		3.6	
	SAF50 10%	0.2	0.2	0.9^{abc}	0.8^{abc}	3.3	3.2
	SAF50 50%	0.1	0.1	0.7^{abc}	0.6^{ab}	2.3	2.3
	SAF113 10%	0.1	0.2	0.9^{abc}	1.1^{abc}	3.3	3.2
	SAF113 50%	0.1	0.1	0.6^{abc}	0.6^{a}	2.3	2.2
	n	8		7		4	
Stat sig.		NS		*		NS	

Different letters in the exponent indicate statistical difference (Tukey HSD) of the scenarios within each group (low, medium or high) at p<0.05 (*) and p<0.001 (***); NS, not significant

When LU-scale results were aggregated to the farm-scale, the only significant reduction occurred at medium erosion sites, where soil erosion was reduced by up to 65% when SAF was combined with contouring over 50% of the farm, on the best quality land, and at both 50 and 113 trees ha^{-1} (Table 4b). A similar effect was expected at high erosion sites. However, the number of samples (n = 4) and the high variability of the results (between 3.4 and 9.7 t ha^{-1} a^{-1}) prevented attainment of statistical significance. Similar relative reductions in erosion rates have been found following introduction of hedgerow intercropping, where soil erosion was reduced by up to 90% on gentle slopes in Nigeria, and by 45-65% on steep slopes in maize systems in Colombia (Young, 1989).

RUSLE does not account for gully erosion. In fact, implementation of SAF without contouring could increase the probability of gully erosion along the tree strips, due to the greater erosivity of water drops under the tree canopy (Young, 1989, McDonald et al., 2003), reducing or negating any positive impact of SAF systems on soil erosion. Nevertheless, correct orientation of the tree lines, along the contour level, would avoid this negative impact (Seobi et al., 2005). Moreover, RUSLE does not model land slide processes, particularly important in slopes with layered clays. Although not modeled, the presence of trees in such areas could lower the risk of land sliding (Sidle et al., 2006).

3.5.2 Nitrogen leaching

The magnitude of nitrate leaching to groundwater strongly depends on the soil water balance. In regions or years of low rainfall, water may not be transported below the root zone because evapotranspiration exceeds rainfall (Lehmann & Schroth, 2003). Such patterns of rainfall, typical of Mediterranean areas, were found at the Spanish LTS (Table 2), where for the arable status quo, nitrogen leaching was at most minimal (Table 3 – ALC and CAM). These results agree with the general observation that leaching from deep soils under rainfed agriculture in Mediterranean climates is negligible (Seligman et al., 1992, Sadras, 2002). For the Atlantic zone, predicted nitrogen leaching in France and the Netherlands for the arable status quo ranged from 37 to 155 kg N ha^{-1} a^{-1} (Table 3). This is similar to reported values of 10 to 80 kg N ha^{-1} for annual nitrogen leaching in rainfed agriculture in temperate European locations (Nemeth, 1996, Ersahin, 2001, Hoffmann & Johnsson, 2003) or slightly higher values of up to 100 kg N ha^{-1} a^{-1} in other temperate locations (Di & Cameron, 2002, Webster et al., 2003). The highest leaching rates were predicted for the LTS in the Netherlands (Table 3). Schröder (1998) reported annual nitrate leaching of 50-250 kg N ha^{-1} in forage maize systems on sandy soils in the Netherlands. The predicted values in Table 3 therefore appear in reasonable agreement with results from the literature.

Scenario comparisons were restricted to the ten French and Dutch test sites where nitrogen leaching exceeded about 10 kg ha^{-1} a^{-1}. The result for the LU (Table 5a) showed a significant reduction in nitrogen leaching by 54% at 113 trees ha^{-1} on the best land. At 50 trees ha^{-1}, the impact of trees on crop yields was smaller and thus the length of the profitable cropping cycle longer, leading to nitrogen application for a longer period. As a result, the predicted reductions of nitrogen leaching were not statistically significant for SAF at 50 trees ha^{-1}.

Table 5: Predicted annual leaching of nitrogen (kg N ha^{-1} a^{-1}) over 60 years under the status quo arable system, and after the introduction of silvoarable agroforestry with 50 (SAF50) or 113 trees ha^{-1} (SAF113), starting with either the best or worst agricultural land, at all sites (>10 kg N ha^{-1}), medium leaching sites (<100 kg N ha^{-1}) and high leaching sites (>100 kg N ha^{-1}) at a) the plot scale (land unit) and b) the farm/landscape scale (landscape test site) on 10% or 50% of the land.

a) Land unit scale	All		Medium (< 100 kg N ha^{-1})		High (> 100 kg N ha^{-1})	
	Worst	Best	Worst	Best	Worst	Best
Status quo	90	109	69	37	142^{ab}	182^{a}
SAF50	85	107	73	44	117^{ab}	171^{ab}
SAF113	70	66	56	34	105^{ab}	99^{b}
N	13 (W=7;B=6)		8 (W=5;B=3)		5 (W=2;B=3)	
Stat. sig.	NS		NS		*	

b) Landscape test site scale	All		Medium (< 100 kg N ha^{-1})		High (> 100 kg N ha^{-1})	
	Worst	Best	Worst	Best	Worst	Best
Status quo	102		53		134^{a}	
SAF50 10%	98	98	54	54	126^{ab}	126^{ab}
SAF50 50%	91	92	58	58	114^{ab}	114^{ab}
SAF113 10%	95	94	53	53	122^{ab}	121^{ab}
SAF113 50%	80	79	52	53	98^{ab}	97^{b}
N	10		4		6	
Stat. sig.	NS		NS		*	

Different letters in the exponent indicate statistical difference (Tukey HSD) of the scenarios within each group (all, medium or high) at p = 0.05 (*); NS, not significant

At farm-scale (Table 5b), differences between scenarios in the level of nitrogen leaching were not statistically significant due to the small number of LTS (n = 10) and the high variability in predicted nitrogen leaching values, ranging from 37 kg ha^{-1} a^{-1} at Champdeniers (CHD) to 155 kg ha^{-1} a^{-1} at Scherpenzeel (SCH). However, when high (>100 kg ha^{-1} a^{-1}) nitrogen leaching sites were analyzed separately, introducing agroforestry at 113 trees ha^{-1} on 50% of the best land of the farm, reduced nitrogen leaching by approximately 30%, from 134 to 97 kg N ha^{-1} a^{-1} (Table 5b).

High nitrogen leaching was generally associated with high crop yields and high fertilizer application rates. At such sites, the YieldSAFE model predicted greater tree-crop competition and the trees therefore reduced crop yield to a larger extent, reducing the annual fertilizer applications, moreover ceased earlier, because intercropping was no longer profitable. These effects were less pronounced at the LTS with currently medium levels of nitrogen leaching, where nitrogen fertilizer application was less intensive and consequently, implementation of SAF did not significantly reduce nitrogen leaching (Table 5b).

The predicted reduction in nitrogen leaching under SAF appears conservative, compared to reported values of 40 and 75%, in temperate agroforestry systems (Udawatta et al., 2002, Nair & Graetz, 2004). However, the modeling approach used here does not account for the potential of tree roots to recover nitrogen from below the crop root zone (Sanchez, 1995, van Noordwijk et al., 1996, Livesly et al., 2000, Rowe et al., 2001) nor the possibility of reducing fertilization due to increase of organic matter in the soil as consequence of tree leaf fall (Thevathasan & Gordon, 2004). In addition, at farm-scale, the predicted nitrogen leaching values are the result of only a 10 and 50% conversion of the total farm area to SAF, with the remainder of the farm under the current arable crops.

3.5.3 Carbon sequestration

Carbon sequestration was calculated for SAF only, since the primary difference in sequestration between arable and SAF systems is due to carbon immobilization in tree biomass (Alegre et al., 2004). Although additional carbon can also be stored in soil due to leaf fall (Dixon, 1995, Montagnini & Nair, 2004) and in the vegetation strip along the tree line, these processes are not included in YieldSAFE and therefore our values can be considered conservative. For the estimates, belowground tree biomass was calculated from the aboveground tree biomass (*see section 3.4.1, page 40*), using allometric relationships described in *section 2.3.4* (page 16). Sequestration varied in dependence of the tree species selected for each LTS. Under the most favorable scenario (113 trees ha^{-1} on 50% in the best quality land) sequestration varied from 0.08 to 0.47 t C ha^{-1} a^{-1} for slow growing trees (holm oak and stone pine), from 0.54 to 0.89 t C ha^{-1} a^{-1} for moderately fast growing trees (wild cherry and walnut), and from 2.1 to 3.0 t C ha^{-1} a^{-1} for fast growing trees (poplar). By year 60, total sequestration was between 5 and 29 t C ha^{-1}, 32 and 54 t C ha^{-1}, and 126 and 179 t C ha^{-1} for slow, moderately fast, and fast growing trees respectively (Table 3). These values are within the range of 3-60 t C ha^{-1} (Kürsten, 2000) or 15-198 t C ha^{-1} (Dixon et al., 1994) for agroforestry systems and 190 t C ha^{-}1 in poplar forests reported for typical tree rotations (van Kooten, 2000, van Kooten et al., 2002, McKenney et al., 2004).

The overall analysis does not show differences between tree densities (Table 6). However, for slow growing trees, carbon sequestration is significantly higher when SAF is implemented in a large portion (50%) of the farm (Table 6b). On the other hand, with medium-fast growing tree species, these significant differences do not occur. These latter results are due to a higher variability of carbon sequestration caused by medium (hybrid walnut and wild cherry) and fast growing trees (poplar). Nevertheless, lower tree densities result in higher biomass per tree (Balandier & Dupraz, 1998, van der Werf et al., 2006), somewhat compensating for the low tree density. No differences in sequestration were found between SAF systems established on high or low quality land, although better results can be consistently found in the best land. At farm-scale, significant differences in sequestration were only found between SAF on 10 or 50% of the farm (Table 6b).

Table 6: Predicted additional carbon sequestration (t C ha^{-1}) after 60 years, relative to that in an arable control, when agroforestry with either 50 (SAF50) or 113 trees ha^{-1}(SAF113) is introduced on the worst or best quality land for slow growing trees (holm oak and stone pine) and medium-fast growing trees (wild cherry, hybrid walnut and poplar) at a) land unit or b) landscape test site scale.

a) Land unit scale	All		Slow growing		Medium-fast growing	
	Worst	Best	Worst	Best	Worst	Best
Status quo	0		0		0	
SAF50	61	45	14^{a}	16^{ab}	81	106
SAF113	67	82	27^{bc}	31^{c}	112	133
N	30		15 (W=8; B=7)		15 (W=7;B=8)	
Stat. sig.	NS		*		NS	

b) Landscape test site scale	All		Slow growing		Medium-fast growing	
	Worst	Best	Worst	Best	Worst	Best
Status quo	0		0		0	
SAF50 10%	7.8^{a}	11.7^{a}	1.7^{a}	1.8^{a}	13.3^{a}	20.7^{ad}
SAF50 50%	28.5^{bc}	43.9^{bc}	8.9^{b}	9.5^{b}	46.1^{bcd}	74.9^{bc}
SAF113 10%	10.7^{a}	15.5^{ac}	3.4^{a}	3.5^{a}	17.2^{a}	26.3^{abd}
SAF113 50%	39.9^{b}	60^{b}	17.3^{c}	18.4^{c}	60.1^{bc}	96.5^{c}
N	19		9		10	
Stat. sig.	*		*		*	

Different letters in the exponent indicate statistical difference (Tukey HSD) of the scenarios within each group (all, slow growing and medium-fast growing) at p = 0.05 (*); NS, not significant. The status quo scenario was not included in the statistical analysis.

3.5.4 Landscape biodiversity

The habitat index (I_{hab}), described by Palma et al. (2005), expresses landscape biodiversity by relating the share of natural and semi-natural habitats to the total area of a given landscape. The introduction of rows of trees in homogeneous arable areas increases the structural diversity of the landscape, which potentially increases its species diversity (Peng et al., 1993, Burgess, 1999, Middleton, 2001, Smart et al., 2002). The effect of SAF on 10 and 50% of the farm was examined assuming that each hypothetical farm was representative of land use in each LTS.

The introduction of SAF increased I_{hab} for each LTS, with the largest increase in areas where existing natural or semi-natural habitat was low. The current I_{hab} of the LTS varied from low values (I_{hab} < 10%) in homogeneous agricultural landscapes (e.g. Table 3 – OCA, CHT, BAL) to high values (I_{hab} > 60%) in more heterogeneous areas (e.g. Table 3 – ALC, ALM, CAR). I_{hab} – values of around 10% increased by a factor of 4 (e.g. CAM), while I_{hab} –values of around 80% increased by about a factor of only 1.15 (e.g. ALC). Mean I_{hab} of all LTS under the status quo was 25%. With 10% of the land under SAF, I_{hab} increased by a factor of 1.28, but significant differences ($p<0.01$) were only found when SAF was implemented on 50% of the farm, increasing I_{hab} by a factor of 2.6 (I_{hab} = 62%).

The habitat index approach can be considered a general and easy method for estimating landscape biodiversity, because it follows the generally accepted principle that landscape heterogeneity favors most taxa (Forman & Godron, 1986). Trees can provide a habitat for some bird, arthropod and small mammal species which otherwise can not inhabit arable landscapes (Peng et al., 1993, Klaa et al., 2005). The grassy or herbaceous strips below the trees consist of either sown plant species or arable weeds. Their contribution to species diversity is equally important, but will depend strongly on management (Griffiths et al., 1998, Burgess et al., 2003); a factor not assessed here. The method does not differentiate between SAF systems at different densities. Subsequent analyses should refine the approach and consider the characteristics and requirements of specific landscapes.

3.5.5 European scale implications

Reisner et al. (2006) identified 90 million hectares (Mha) of European arable land potentially suitable for SAF systems using hybrid walnut, wild cherry, poplar, holm oak and stone pine. Within this area, the study identified 65 Mha where SAF could potentially reduce soil erosion and nitrogen leaching, and increase landscape biodiversity. This paper has investigated to what extent those environmental benefits could be realized through SAF systems and what they could contribute to carbon sequestration.

Eight million hectares of European arable land are seriously threatened by erosion (Reisner et al., 2006) and SAF, with one of the five tree species examined here, could potentially be implemented on 2.6 Mha of this land (Reisner et al., 2006). If farmers in these areas would combine SAF with contouring on the best 50% of their farm land, soil erosion could be reduced by as much as 65%.

Nitrogen leaching could be reduced on 12 Mha of land (Reisner et al., 2006) through use of SAF, mainly in central and northern Europe. These reductions could potentially be as high as

28%, if SAF was implemented at high densities (113 trees ha^{-1}) on 50% of the best farm land. In addition, nitrogen uptake below the root zone of annual crops might further reduce nitrogen leaching at these sites, although this has not been considered here and requires future investigations.

Carbon sequestration could also be increased on the 90 Mha of European arable land potentially suitable for SAF(Reisner et al., 2006). As tree density and land quality did not significantly affect cumulative sequestration, carbon sequestration could be maximized by maximizing the area of land converted to SAF. The use of medium-fast growing tree species in SAF systems when implemented on 50% of the agricultural land could contribute 0.77-1.6 t C $ha^{-1}a^{-1}$ (46-96 t C ha^{-1}) to sequestration over a 60-year period. However, values up to 3 t C $ha^{-1}a^{-1}$ (179 t C ha^{-1}) are potentially feasible.

Our assessment of potential carbon sequestration differs from that of the European Climate Change Program (ECCP), which estimates that less than one million hectare of land in Europe is suitable for agroforestry, and that no net change in annual carbon balance will occur by 2010 (ECCP, 2003). However, that represents a medium-term perspective. The actual adoption of SAF will depend on both, its profitability and its legal status, which could change in the coming years, stimulating the uptake of SAF systems by European farmers (Lawson et al., 2004, Lawson et al., 2005).

Monotonous arable landscapes, defined by Reisner et al. (2006)as areas where arable land covers over 50% of the total land area in a 25 km^2 area, cover about 100 Mha in Europe(Reisner et al., 2006). Approximately 21 Mha of this land would be suitable for SAF using one of the five tree species tested here, which could significantly increase landscape biodiversity. This broad assessment, however, needs further refinement by taking into account specific landscape characteristics on the one hand and target species on the other hand. Baldi et al. (2005) and Stoate et al. (2003) have shown that it is impossible to design a management scheme that favors all species. Moreover, for some steppic wildlife species, open rather than structured landscapes are required (e.g. *Otis tarda* L.), and some regions, such as Brandenburg (northern Germany) are traditionally characterized by large open fields with specific combinations of fauna and flora.

3.6 Conclusions

A modeling approach was developed and used in randomly selected LTS for an initial assessment of the potential environmental benefits of SAF at European scale.

The results for 19 randomly selected LTS in Spain, France, and the Netherlands showed that adoption of silvoarable agroforestry systems can potentially lead to reduced soil erosion and nitrogen leaching and increased carbon sequestration and landscape biodiversity. The extent of the modifications depends on the problems associated with each site and the management of the SAF system selected for each location. Predicted environmental benefits were highest when SAF was implemented on large areas (i.e. 50% of the farm) on high quality land, where current agricultural practices are most intensive and thus associated with higher levels of soil erosion and nitrogen leaching. Tree density (50 or 113 trees ha^{-1}) appears less important, as in stands of lower density biomass production per tree is higher, reducing the difference in values of the indicators on an area basis. The reduction in nitrogen leaching, however, was stronger at high tree densities.

Further research is needed to improve the environmental modeling approach of SAF systems in order to provide answers to relevant questions of stakeholders. For example, how is nitrogen leaching affected in Mediterranean irrigated systems? What would be the optimum tree density? What would be the best field arrangement of tree lines for a given density?

Agroforestry systems are highly diverse. We only examined five tree species in combination with five crops but, of course, many more tree species and crop types can and should be considered. Their choice and the manifold possible layouts of the system with respect to the density and arrangement need to be adapted to local conditions and farmer's preferences. All those options can only be fully explored with modeling approaches. Such models, however, must be validated on the basis of experimental data from these systems and such data are scarce.

In Europe, new land-use systems should (also) yield environmental benefits. The results presented here increase our understanding of the environmental benefits that can be expected from modern agroforestry systems and complement an economic analysis of such systems in Chapter 4 for the same LTSs. In further research, an integrated economic and environmental analysis of the benefits provided by SAF systems will be presented (Chapter 5).

Chapter 4

The development and application of bio-economic modelling for silvoarable systems in Europe

Based on Graves AR, Burgess PJ, Palma JHN, Herzog F, Moreno G, Bertomeu M, Dupraz C, Liagre F, Keesman K, van den Werf W, de Nooy AK, van den Briel JP, 2006, The development and application of bio-economic modelling for silvoarable systems in Europe, Ecological Engineering, accepted

4.1 Abstract

The European Union has introduced measures to promote the integration of trees within farm businesses. Although silvoarable agroforestry is one method by which this can be achieved, the implications at a plot- and farm-scale are poorly understood. From 2001 to 2005, the Silvoarable Agroforestry for Europe project therefore developed computer-based tools to evaluate both the biophysical and economic performance of arable, forestry and silvoarable systems under different European conditions. A biophysical model called "Yield-SAFE", based on light and water competition, was developed to predict long-term arable, forestry and silvoarable yields for given sets of climate and soil conditions. The output from this model was then used in a plot- and farm-scale economic model called "Farm-SAFE" to determine profitability and resource use. Both models were parameterised and used for selected regions of France, Spain and the Netherlands. The analysis in France suggests that walnut and poplar silvoarable systems could provide a profitable alternative to arable and forestry systems, while in Spain a modest restructuring of the amount and delivery of agricultural payments would increase the attractiveness of silvoarable systems of holm oak and stone pine. In the Netherlands, low timber value and the opportunity cost of losing arable land for slurry manure application made both silvoarable and forestry systems uncompetitive with arable systems.

4.2 Introduction

Agroforestry is a form of multi-cropping involving at least one woody-perennial species and significant ecological and economic interactions. Agroforestry systems can be described by their components (crops, animals and trees) and their spatial (dispersed or zoned) and temporal (coincident to sequential) arrangement (Nair, 1985, Sinclair, 1999). Silvoarable agroforestry, defined as the practice of growing an arable crop between spatially-zoned trees (Dupraz & Newman, 1997, Burgess *et al.*, 2004b), is a form of agroforestry that could be undertaken on mechanised arable farms in Europe.

The majority of research on agroforestry systems has been undertaken to evaluate their biophysical performance despite the observation that it is often socio-economic constraints that limit their adoption (Mercer *et al.*, 1998, Graves *et al.*, 2004). Since there are potentially many biophysical and socio-economic interactions between the tree and crop components of silvoarable systems (Dyack et al., 1998) there is a need to consider both the biophysical and socio-economic aspects together. However both the biophysical and the socio-economic analysis of such systems are constrained by lack of experimental data describing the effect of different permutations, for example spacing and different tree species. There are also problems in describing the socio-economic integration and the interaction between the short- and long-term components over the length of a tree rotation.

Computer simulations provide a means of systematically undertaking biophysical and economic analyses of silvoarable systems in the absence of empirical data. Various biophysical and economic models have been developed for monocultures of arable and forestry systems, but few have been developed for silvoarable agroforestry (Graves et al., 2005a). The current bio-economic models of silvoarable systems range from detailed biophysical models with limited economic analysis to economic models that use biophysical data from an external source (Graves et al., 2005a). Bio-economic models have been used to examine the profitability (Thomas, 1991, Willis *et al.*, 1993, Thomas & Willis, 1997, Burgess *et al.*, 2000) and feasibility (Dupraz *et al.*, 1995) of silvoarable systems in Europe. Profitability is normally assessed at a one-hectare scale and performance is compared to competing enterprises such as arable agriculture and forestry. Feasibility is often determined at a farm-scale to view how silvoarable agroforestry affects cash-flow and resource use. This Chapter describes the integrated use of a bio-physical and an economic model, at both a one-hectare- and farm-scale, to determine the potential profitability and feasibility of silvoarable agroforestry in Europe.

4.3 Method

The study focused on three countries (Spain, France and the Netherlands) with differing climates, tree and crop species, and levels of practical experience in implementing agroforestry. Potential sites for the uptake of silvoarable agroforestry, termed landscape test sites, were identified in each country using a geographical information system. Annual yields of trees and crops were derived using a bio-physical model called “Yield-SAFE” (van der Werf et al., 2006) and profitability and feasibility were determined using an economic model called “Farm-SAFE”(Graves et al., 2006).

4.3.1 Identification and Characterisation of landscape test sites

As this Chapter was coordinated with the previous, the landscape test sites were the same as in Chapter 3 (*see section 2.3.1, page 11*).

The land available for silvoarable agroforestry in each landscape test site was assumed to be equivalent to the land available for arable production and this was selected by excluding non-arable land. The area of a specialist cereal farm in each landscape test site was determined from regional data in the Farm Accountancy Data Network (FADN) (EC, 2003) in Spain, the Agricultural Economics Research Institute in the Netherlands (AERS, 2005) and from the Réseau d'observation des systèmes d'exploitation (ROSACE) (APCA, 2005) in France. Where there were no data relating to a specialist cereal farm, farm size was related to the most frequently occurring farm types for the landscape test site region, in the case of Alcala la Real in Spain, an olive farm, and in the case of the Netherlands, pig, dairying and general field cropping farms. At each landscape test site, the proportion of the area of each land unit relative to the total area of the land units was used to represent the proportion of each land unit within a hypothetical farm.

4.3.2 Selection and management of tree and crop species

Annual yields of trees and crops in arable, forestry and silvoarable systems were required for each land unit as inputs for the economic analysis. The tree and crop species for forestry and arable production were chosen to reflect the most the likely practice at each landscape test site. In France and the Netherlands, the trees were selected because they were timber trees; in Spain the choice of tree also reflected policy constraints and issues of ecological importance. The forestry systems selected for Spain comprised holm oak (*Quercus ilex*) and stone pine (*Pinus pinea*). In France wild cherry (*Prunus avium*), walnut (*Juglans* spp.), and poplar (*Populus* spp.) were chosen and walnut and poplar were selected in the Netherlands. The arable systems in Spain were based on wheat, sunflower and fallow. In Poitou Charentes and Centre in France, they were based on wheat and sunflower and in Franche Comté, on wheat, oilseed and grain maize; in the Netherlands on wheat and forage maize. The silvoarable systems integrated the forestry tree species and arable crop species and rotation for each land unit.

The management of the forestry systems at each landscape test site was based on local practice. In Spain, planting densities, thinning and pruning for oak were derived from Pulido et al. (2003) and for stone pine from Yagüe (1994) and Montero and Cañella (2000). In France management for forestry systems was developed from the Institut pour le Développement Forestier (IDF, 1997), Souleres (1992), Boulet-Gercourt (1997) and the Centre Régional de la Propriété Forestière (CRPF, 1997) for walnut, wild cherry and poplar. In the Netherlands, the receipt of grants was conditional on an appropriate planting density, given by the Ministerie van Landbouw, Natuur en Voedselkwaliteit (MLNV, 2004), and thinning and pruning regimes were applied using the management rules in France. The management for the arable systems reflected local practice.

4.3.3 Biophysical modelling

The radiation, temperature, rainfall, soil depth and texture data for each land unit were used as inputs in a daily time-step bio-physical model of tree and crop production, based on competition for light and water (Yield-SAFE) (van der Werf et al., 2006) and implemented in Microsoft Excel© by Burgess *et al.* (2004a) to predict annual tree and crop yields.

The parameters used in Yield-SAFE to describe the growth of each tree and crop species were determined from published material and calibrations. An initial calibration for "potential" monoculture yields (van Ittersum & Rabbinge, 1997) was undertaken against datasets of tree volume and crop yields under high yielding conditions in the Atlantic and Mediterranean zones, assuming within the model that light and temperature but not water, limited growth (Burgess *et al.*, 2004a). Then at each landscape test site and assuming light, temperature, and water limited growth within the model, the values of three parameters (harvest index, water use efficiency and a management factor) were adjusted within acceptable boundaries so that output from the model over the duration of the tree component matched an "actual" monoculture tree and crop yield (van Ittersum & Rabbinge, 1997). The tree and crop management defined previously for the monocultures and "reference" soil depth and texture were also used. The monoculture management and actual and reference values were determined for each landscape test site during workshops held in each country (Herzog *et al.*, 2004, Palma & Reisner, 2004, Reisner, 2004).

In Spain, the actual timber volumes for oak and stone pine in all the landscape test sites in year 60 were assumed to be 0.22 m3 and 0.26 m3 tree-1 respectively, indicating slow growth. In France, wild cherry (1.04-1.06 m^3 $tree^{-1}$) and walnut (1.04 m^3 $tree^{-1}$) for the same rotation were comparatively fast-growing trees. Poplar was the fastest growing tree with actual yields of 1.46-1.51 m^3 $tree^{-1}$ after 20 years. In Spain, actual yields for wheat were comparatively low (1.62-3.71

t ha^{-1}) compared to those in France (6.5-8.0 t ha^{-1}) and the Netherlands (7.8 t ha^{-1}). Actual sunflower yields were lower in Spain (0.60-1.09 t ha^{-1}) than in France (2.3-2.5 t ha^{-1}). Actual yields for oilseed (3.2-4.0 t ha^{-1}) and grain maize (7.5-8.0 t ha^{-1}) were assumed only for France and an actual yield for fodder maize (12 t ha^{-1}) assumed only for the Netherlands.

Using the parameter set developed for actual yields and soils at each landscape test site, tree and crop yields for each land unit were predicted for monoculture forestry and arable systems and two silvoarable systems of 50 or 113 trees ha^{-1}. From the biophysical yields, it was possible to estimate a land equivalent ratio (*LER*) for each system. LERs were initially defined for mixed cropping systems (Mead & Willey, 1980) and have been adapted for agroforestry systems (Ong, 1996, Dupraz, 1998). The LER is "the ratio of the area under sole cropping to the area under the agroforestry system, at the same level of management that gives an equal amount of yield" (Ong, 1996) and is expressed as:

$$LER = \frac{Tree\ silvoarable\ yield}{Tree\ monoculture\ yield} + \frac{Crop\ silvoarable\ yield}{Crop\ monoculture\ yield} \qquad \text{(Equation 13)}$$

Where more than one crop occurred in the rotation, a weighted ratio for each crop was used, depending on its proportion in the rotation.

4.3.4 Economic modelling

The predicted annual yields of trees and crops were used as inputs for a plot- and farm-scale cost-benefit economic model called "Farm-SAFE" (Graves et al., 2006). In arable systems, profitability is typically compared on an annual and per unit area basis by adding the revenue generated (*R*) to the variable costs associated with generating that revenue (*V*) to give a gross margin (Gross margin = $R - V$) (MAFF, 1983, Nix, 1999). However, in tree-based systems, "assignable fixed costs" such as labour and machinery (*A*) are commonly included and can be derived per unit area. Therefore the arable, forestry, and silvoarable systems were compared using their net margin (Net margin = $R - V - A$) (Willis et al., 1993, Burgess et al., 2000, Graves et al., 2006). As the benefits and costs associated with tree-based systems occur over many years, discounted cost benefit analysis was used to define the "present" value of future costs and benefits from the arable, forestry and silvoarable systems using the approach defined by Faustmann (1849). The net "present" value (*NPV*; units: € ha^{-1}) was expressed as:

$$NPV = \sum_{t=0}^{t=T} \frac{(R_t - V_t - A_t)}{(1+i)^t} \qquad \textit{(Equation 14)}$$

Where: *NPV* was the net present value of the arable, forestry or silvoarable enterprise (€ ha^{-1}), R_t was the revenue from the enterprise (including subsidies) in year *t* (€ ha^{-1}), V_t was the variable costs in year *t* (€ ha^{-1}), A_t was the assignable fixed costs in year *t* (€ ha^{-1}), *T* was the time horizon (years), and *i* was the discount rate (discount rate = 4%).

In order to compare systems with different rotation lengths, an infinite net present value was calculated. This was the net present value defined over an infinite rotation, in which each replication had a rotation of *n* years. The infinite *NPV* was defined as:

$$\textit{Infinite NPV} = NPV \frac{(1+i)^n}{(1+i)^n - 1} \qquad \textit{(Equation 15)}$$

The infinite net present value was also expressed as an equivalent annual value (*EAV*) using the following formula:

$$EAV = \textit{infinite NPV} \times i \qquad \textit{(Equation 16)}$$

Assessing the feasibility of a given system involves determining how it modifies flows of farm resources. This is achieved by multiplying plot-scale flows of money, land, and labour by their area on the farm and aggregating the results, then substituting a given system with another system, and assessing the effect on farm resources with and without the substituted system. A maximum of four arable, four forestry and four silvoarable systems could be used to represent a single farm in the "Farm-SAFE" economic model. Economic feasibility was determined using the infinite *NPV* of the farm ($iNPV_{farm}$; units: € farm^{-1}). This combined the *NPV* of the different systems and the *NPV* of "farm fixed costs" (F_t: units: € farm^{-1}) over the same period of time and was defined as:

$$iNPV_{farm} = \left(\sum_{l=1}^{l=4} \left(NPV_a a_a + NPV_f a_f + NPV_s a_s \right) - \sum_{t=0}^{t=T} \frac{F_t}{(1+i)^t} \right) \frac{(1+i)^n}{(1+i)^n - 1} \qquad \textit{(Equation 17)}$$

Where: l was one of four possible land units, NPV_a, NPV_f, and NPV_s were the net present values (€ ha^{-1}) of arable, forestry and silvoarable enterprises in each land unit l; a_a, a_f, and, a_s were the area (ha) of arable, forestry, and silvoarable systems in each land unit l, F_t was the farm fixed cost in year t (€ farm^{-1}), T was the time horizon (years), i was the discount rate and n was the duration of the rotation (years).

4.3.5 Parameterisation and use of Farm-SAFE

The financial data for arable, forestry and silvoarable systems were collected on electronic templates for each landscape test site, using local and national statistics, and expert opinion.

4.3.5.1 Arable and crop component finance

The revenue (crop value and associated subsidy), and the variable and assignable fixed costs for each arable system are described fully by Graves *et al.* (2005b). However, for clarity some key values are described. The assumed value of the arable crops ranged from 85 € t^{-1} for grain maize to 280 € t^{-1} for sunflower; the assumed value of wheat grain ranged from 102 to 142 € t^{-1}. Assumed variable costs tended to be lowest in Spain (45-189 € ha^{-1}) and highest in the Netherlands (457-479 € ha^{-1}), and assignable fixed costs such as machinery and labour followed a similar pattern. For the crop component of the silvoarable system, the variable and assignable fixed costs were applied according to the proportion of intercrop area in the system which was constant. Also, as intercrop yields decrease over time due to tree growth, it was assumed that cropping would only continue for as long as the intercrop net margin (calculated on a five year moving average to remove the effect of yield failure caused by poor weather) was profitable, after which it was assumed the intercrop area would be fallow.

4.3.5.2 Forestry and tree component finance

The financial data for forestry and the tree component of the silvoarable system comprised the revenue from timber and subsidies, and the costs of woodland establishment and management. These are summarised below, but explained fully in Graves *et al.* (2005b). The revenue from timber was calculated using relationships between the standing value of the tree and the average tree volume for each species in each country. In Spain, the value of oak (17 € m^{-3}) and pine (8-19 € m^{-3}) was low. By contrast, in France, the value of walnut (40-1300 € m^{-3}), wild cherry (10-380 € m^{-3}), and poplar (7-55 € m^{-3}) was relatively high; thinned timber, given a different per cubic metre price to clear-felled timber, was also relatively valuable. In the Netherlands, the perceived value of walnut (18-41 € m^{-3}) was much lower than in France, but the value of poplar (19-97 € m^{-3}) was slightly higher.

The costs associated with the forestry system and the tree component of the silvoarable systems were based on numerous sources. Costs varied between countries, tree species and regions and regarding the tree component of the silvoarable system, were not assumed to be proportional to the number of trees or the area of the tree component (except in the Netherlands), as was the assumption for the crop component. The cost of ground preparation was anticipated to be highest in Spain and the Netherlands and lowest in France. This was due to difficult soil conditions in Spain, where it was anticipated that tree pits would need to be prepared, requiring use of specialised machinery (including labour) at a contract rate of 31 € hr^{-1}. In the Netherlands, it was anticipated that labour and machinery would be provided by external enterprises at a cost of 22 € hr^{-1}. In France, however, it was anticipated that the farmer would undertake the majority of operations at a cost of € 7.8 hr^{-1}. The cost of planting materials was greatest for walnut (6 € $tree^{-1}$) and poplar (4 € $tree^{-1}$) in France and walnut (5 € $tree^{-1}$) in the Netherlands. Oak (0.36 € ha^{-1}) and pinus (0.76 € ha^{-1}) in Spain were relatively inexpensive. Tree protection materials, such as spiral guards or fencing, were highest for walnut and cherry in France (1.5 € $tree^{-1}$) and lowest for walnut and poplar in the Netherlands (0.29 € $tree^{-1}$). The time required for planting and protecting the trees was highest in Spain (2.7 min $tree^{-1}$), than France (1.0-2.0 min $tree^{-1}$), and lowest in the Netherlands (0.8 min $tree^{-1}$). In France (15 € hr^{-1}) and the Netherlands (22 € hr^{-1}), it was anticipated that planting and protection would be carried out by externally contracted enterprises; in Spain it was anticipated that this would be done using locally available labour (7.8 € hr^{-1}). The full establishment cost of forestry systems was greatest in the Netherlands (3420 € ha^{-1} for walnut; 1940 € ha^{-1} for poplar) and lowest in Spain (770 € ha^{-1}) for oak systems at 400 trees ha^{-1}. The full establishment cost for forestry systems of cherry (1510 € ha^{-1}), walnut (1633 € ha^{-1}), and poplar (1260 € ha^{-1}) in France and high density oak (1470 € ha^{-1}) and pine (1786 € ha^{-1}) in Spain were between these extremes. The full establishment cost of the tree component in the silvoarable systems was lower for each species. For the 113 trees ha^{-1} systems, these ranged from 1200 € ha^{-1} for walnut in the Netherlands to 233 € ha^{-1} for oak in Spain; for the 50 trees ha^{-1} systems they ranged from 710 € ha^{-1} in the Netherlands to 120 € ha^{-1} for oak in Spain.

Significant maintenance costs included weeding, sward establishment, pruning and thinning. In Spain, it was anticipated that management would be minimal because of the low financial value of the oak and pine timber. The main costs in the forestry system were associated with weeding in the initial three years and establishing a grass sward in year 12. For the tree component of the silvoarable system, the only cost-bearing maintenance operation was assumed to be weeding in the initial five years. Both these operations were assumed to be externally contracted at a rate of 31 € h^{-1}. Pruning and thinning were assumed to be free of cost as an established system exists whereby harvested oak and pine timber is given in lieu of payment to those who undertake the work. By contrast, management was much more intensive in France and in the Netherlands. In France, the control of undergrowth between trees was a significant cost for about the first quarter of a forestry rotation and for the duration of arable cropping in a silvoarable system. Other significant costs included pruning and an annual land tax that varied marginally between regions. In the Netherlands, the costs of establishing a grass sward in the first year (417 € ha^{-1} grass) and subsequent maintenance (136 € ha^{-1} grass a^{-1}) were high. Pruning, especially for walnut, and thinning were also significant costs. In addition, it was assumed that an opportunity cost (a nitrate levy of 408 € $ha^{-1}a^{-1}$) was incurred when arable land was converted to forest, because the land could no longer be used to accept slurry manure. This was also applied on a pro-rata basis to the tree-strips in the silvoarable system.

4.3.5.3 Pre-2005 grant regime

The relative profitability of forestry, arable and agroforestry systems on farms in the The relative profitability of forestry, arable and agroforestry systems on farms in the European Union is significantly affected by the grant regime. In the pre-2005 grant scenario, it was assumed that direct payments on the arable system and crop component of the silvoarable system would be dependent on the crop species and the portion of arable land in the system. These were greatest for maize (400 € ha^{-1}) in the Netherlands and least for wheat (129 € ha^{-1}) in Spain, but also varied with crop species and in France, with region. The pre-2005 payments on forestry and tree component of the silvoarable systems were established from local and national statistics and expert opinion. In Spain, farmers received a planting grant (849-1593 € ha^{-1}) dependent on tree species, a compensation payment (225-325 € ha^{-1} a^{-1}) for 20 years depending on location and previous land-use and a maintenance grant (180-288 € ha^{-1} a^{-1}) for five years, subject to appropriate management of the trees (Graves et al., 2005b). In France, in Poitou Charentes and Centre, planting grants covered 50% of tree costs in the first four years and compensation payments (240-300 € ha^{-1} a^{-1}) were available for walnut and cherry for ten years and for poplar for seven years. In Franche Comté, there were no grants or payments, due to existing and substantial areas of forest. In the Netherlands, a planting grant of 95% of costs was available up to a maximum of 1500 € ha^{-1}, a compensation payment of 100 € ha^{-1} a^{-1} for five years and a maintenance payment of 545 € ha^{-1} a^{-1} for 18 years. For the tree component of the silvoarable system, all tree payments were forfeited in Spain and the Netherlands. In the Poitou Charentes and Centre regions of France, establishment grants were available at 50% of the tree costs in the first four years, but no tree payments were available in Franche Comté.

4.3.5.4 Post-2005 grant regime

In the post-2005 grant scenario, the changes anticipated for the Common Agriculture Policy were implemented. For the arable crop, the changes meant that the area payments could be fully decoupled from crop type, resulting in a single farm payment for as long as the land was cropped. The per hectare value of these payments were calculated to be lowest in Spain (116-330 € ha^{-1}) and highest in France (329-353 € ha^{-1}) and the Netherlands (353-586 € ha^{-1}). In the post-2005 scenario for forestry, existing levels of payments applied, where they were in accordance with the rural development strategy of the European Union (EC, 2004b). In France, there was therefore no change, but in Spain and the Netherlands, planting payments at each site were changed to 50% of tree costs in the first four years. The compensation payments and maintenance grants were reduced to 500 € ha^{-1} a^{-1} with a maximum duration of 10 years, unless they were already below these levels. In that case existing values were used.

Since the effect of these changes on silvoarable systems is still unclear, two extreme scenarios were developed for the post-2005 situation (Table 7). In scenario 1, the single farm payment was assumed for the percentage of crop area in the system with no tree payments. In scenario 2, the single farm payment was assumed for the whole system with 50% of the tree costs in the first four years covered by a planting grant.

Table 7: Two extreme post-2005 grant scenarios assumed for silvoarable agroforestry

	Arable payment	Tree payment
Scenario 1	Percentage crop area in system	None
Scenario 2	Total area of system	Fifty percent costs in years 1-4

4.3.6 Farm-scale data

Only a brief description of the approach and data used in the farm-scale modelling is provided here. A more detailed description can be found in Graves et al. (2005a).

Economic feasibility was assessed by multiplying the one-hectare results for each land unit by their area and adding farm fixed costs from the FADN and ROSACE for the hypothetical farms at each site (Equation 17). The quality of the land units was ranked assuming that higher average yields meant better land. Expert opinion was then used to determine which tree species and which crop rotation would be most suitable for each land unit. The infinite net present value of the farm was used to evaluate the economic effect of planting 10% of the farm with forestry or silvoarable systems in comparison with the status quo arable farm under the pre-2005 and post-2005 grant regimes. Planting was assumed in year 1 and holm oak, stone pine, wild cherry and walnut were "harvested" to provide revenue in year 60. A rotation of 20 years was assumed for poplar, and by re-planting in years 21 and 41, three full rotations of poplar were completed in 60 years. It was assumed for poplar that the tree related grants in year 21 and 41 would be the same as for year 1.

4.4 Results and discussion

4.4.1 Biophysical production in arable and forestry systems

The predicted yield of the monoculture arable crops within a specific year on the 42 land units ranged from 0.2 t ha^{-1} for sunflower in Spain to 15.9 t ha^{-1} for maize in the Netherlands (Table 8). Although the greatest absolute variation in yield was associated with high yielding crops in the Netherlands and France, the relative variation in yields was greatest in Spain. For the forestry systems, the mean timber volume per tree ranged from 0.25 m^3 for stone pine after 60 years, to 1.34 m^3 for poplar after 20 years. The maximum recorded tree size was for poplar (1.59

m^3) in France and the minimum for oak (0.23 m^3) in Spain. The standard deviation suggested that absolute variation was greatest for wild cherry in France and poplar in the Netherlands. The coefficient of variation showed that the relative variation was greatest for wild cherry, oak and poplar in the Netherlands.

Table 8: Summary and description of yields for crops and trees in France, Spain and the Netherlands

Country	Arable crop	Count	Mean ($t\ ha^{-1}$)	Standard deviation ($t\ ha^{-1}$)	Range ($t\ ha^{-1}$)	Coefficient of variation (%)
Spain	Sunflower	120	0.8	0.4	0.2-1.7	52
	Wheat	697	2.5	1.0	0.6-5.8	40
France	Grain maize	61	6.3	1.2	2.9-9.8	20
	Oilseed	260	3.2	0.4	1.9-4.3	13
	Sunflower	106	1.7	0.4	0.7-2.6	26
	Wheat	613	5.5	1.5	0.9-10.5	27
the Netherlands	Forage maize	80	11.5	1.7	8.0-15.9	15
	Wheat	20	7.9	1.2	5.9-11.1	16
	Tree species		($m^3\ ha^{-1}$)	($m^3\ ha^{-1}$)	($m^3\ ha^{-1}$)	(%)
Spain	Oak (60)	16	0.33	0.050	0.23-0.43	15
	Pine (60)	3	0.25	0.005	0.25-0.26	2
France	Cherry (60)	12	0.88	0.151	0.71-1.15	17
	Poplar (60)	4	1.34	0.143	1.26-1.59	11
	Walnut (60)	4	1.01	0.008	1.00-1.02	1
the Netherlands	Poplar (20)	2	1.28	0.215	1.06-1.49	17
	Walnut (60)	1	0.71	n/a	0.71	n/a

Note: values in brackets show length of rotation

Within each landscape test site, crop yield within a land unit could potentially vary with soil depth, soil type and radiation level. For each crop, except wheat in Spain, there was a significant positive correlation between predicted annual crop yields and soil depth (Table 9). The standard error of the estimate showed that in absolute terms, variation was greatest for wheat in Spain and France. However, in relative terms, the variation was greatest for wheat in Spain. Predicted timber yields were also positively correlated with soil depth for cherry, poplar and oak (Table 9). However, this correlation was only significant ($P=0.05$) in the case of wild cherry.

In each country, analysis of variance (analysis not summarised here) showed that there were significant differences (P=0.05) in soil texture and predicted crop yields, except in the case of oilseed in France. However, there were no significant difference in soil texture and predicted timber yields in any of the countries.

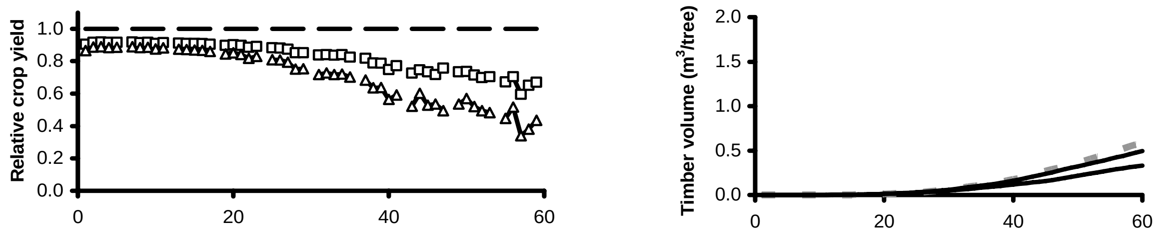
Table 9: Relationship between a) crop yield and b) timber volume and soil depth for selected crop and tree species in Spain and France

		Count	Regression of y against depth (*d*)	Correlation coefficient	Significant (P=0.05)	Standard error of estimate	Confidence interval
	Crop						
Spain	Wheat	697	0.00019 *d* + 2.34	0.02	No	1.0200	2.0000
France	Wheat	613	0.0215 *d* + 3.69	0.57	Yes	1.2300	2.4200
	Grain maize	61	0.0182 *d* + 4.90	0.67	Yes	0.9200	1.8500
	Sunflower	106	0.0072 *d* + 1.06	0.49	Yes	0.3900	0.7700
	Oilseed	260	0.0032 *d* + 2.95	0.26	Yes	0.4200	0.8300
	Tree species						
Spain	Oak	16	0.0012 *d* +0.169	0.53	No	0.044	0.094
France	Cherry	12	0.0029 *d* + 0.6451	0.75	Yes	0.105	0.233
	Walnut	4	0.000028 *d* +1.02	-0.12	No	0.01	0.041
	Poplar	4	0.0042 *d* + 0.984	0.97	No	0.041	0.177

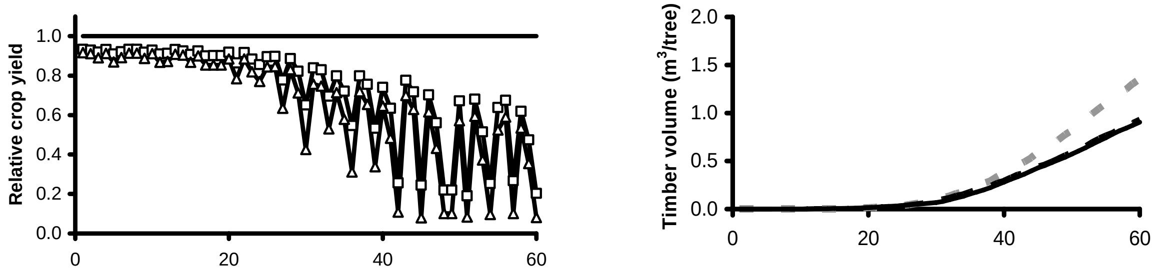
4.4.2 Biophysical production in silvoarable systems

The biophysical outputs from Yield-SAFE for the silvoarable systems (50 and 113 trees ha^{-1}) showed a general decline in crop yields as the trees became larger and competed more effectively for light and water. Typical relations for four land units are shown in Figure 12. Oak (Figure 12a) and stone pine (which showed similar growth over time to oak and is therefore not shown) grew slowly throughout the whole rotation. Hence relatively high crop yields were sustained for most of the tree rotation. The initial rate of timber formation by wild cherry (Figure 12b) was slow compared with walnut (Figure 12c) and poplar (Figure 12d), and crop yield reduction in the walnut and poplar systems was predicted to occur earlier than in the wild cherry systems. The model was also used to predict difference in crop and tree yield at two tree densities (50 and 113 trees ha^{-1}). As expected relative crop yields were greatest in the 50 tree ha^{-1} system and relative timber yields (m^3 ha^{-1}) were greatest in the 113 tree ha^{-1} system (Figure 13).

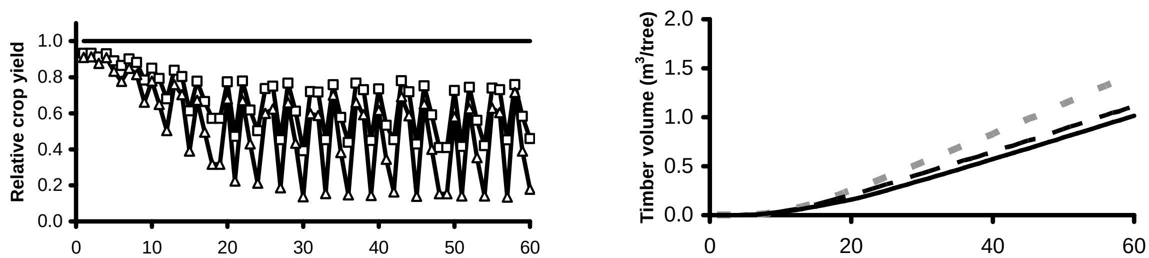
a) Land unit 2, St Maria del Campo, Spain (oak; wheat/wheat/wheat/wheat/wheat/fallow)



b) Land unit 1, Champdeniers, France (wild cherry; wheat/wheat/s/wheat/oilseed/sunflower)



c) Land unit 2, Champdeniers, France (walnut; wheat/wheat/s/wheat/oilseed/sunflower)



d) Land unit 1, Sherpenzeel, the Netherlands (poplar; forage maize)

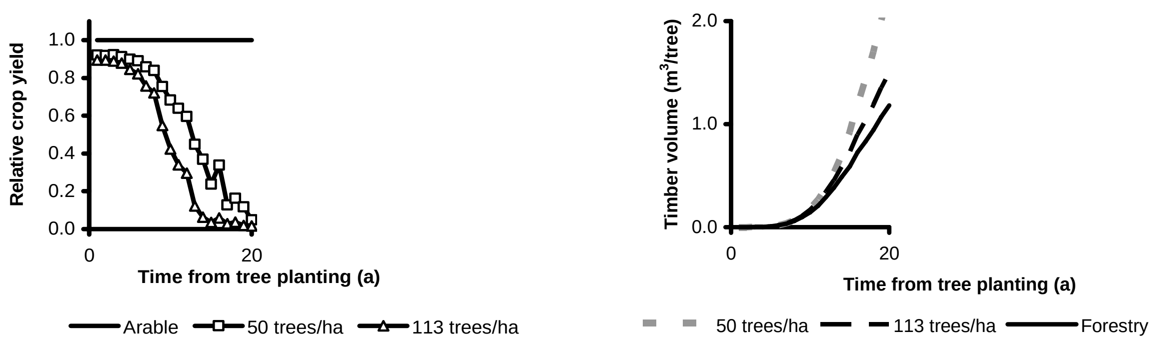


Figure 12:* *Relative crop yields and the timber volume for (a) an oak, b) a wild cherry, c) a walnut and a d) poplar silvoarable agroforestry system for selected land units

a) Relative yields for 113 trees ha^{-1} **b) Relative yields for 50 trees ha^{-1}**

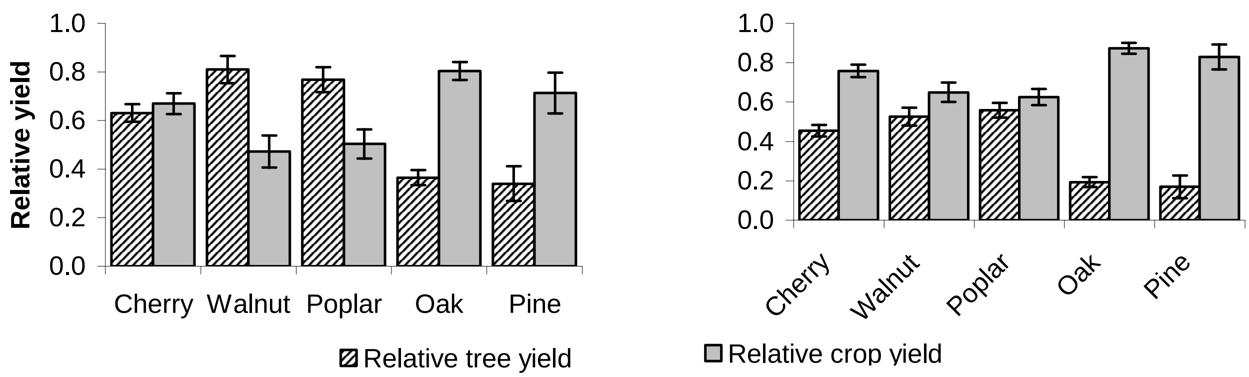


Figure 13: Predicted effects of tree species in a silvoarable system planted at a) 113 trees ha^{-1} and b) 50 trees ha^{-1} on the yield of the tree and the crop components relative to a monoculture (error bars show confidence intervals for mean values)

In Spain, the relative yields of autumn-planted species, such as wheat, tended to be greater than for spring-planted crops, such as sunflower (Figure 14a). As oak and stone pine are evergreen species, it was assumed that this was due to greater competition experienced by the spring-planted crop for water. In France, the difference in the relative yield of the autumn- (i.e. wheat and oilseed) and spring-planted (sunflower and grain maize) crops was larger than in Spain (Figure 14b). This was probably due to reduced competition for light, because the tree species planted in France were deciduous and hence had no leaves for a large proportion of the growing period of the autumn-planted crops, whereas in Spain as the trees were evergreen and competition for light was similar for both the spring and autumn-planted crops. Under poplar in the Netherlands (Figure 14c), similar effects regarding the difference between autumn-planted wheat and spring-planted forage maize were evident. These patterns were similar in both the low density and high density systems, but the relative yields of the crops were higher at 50 trees ha^{-1} than at 113 trees ha^{-1}.

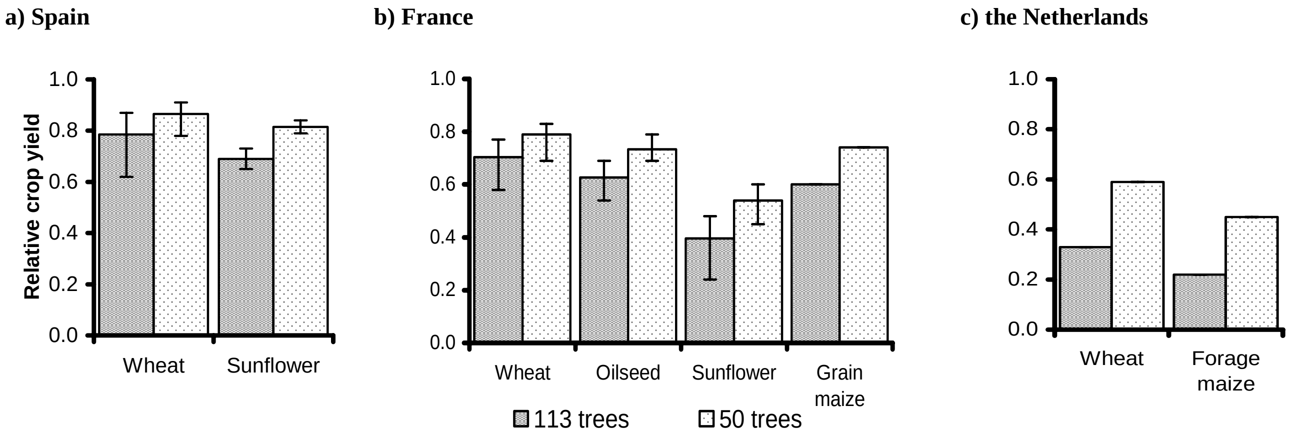


Figure 14: Effect of crop species on the relative crop yield over a complete tree rotation, in (a) Spain and (b) France, under all tree species and in (c) the Netherlands under poplar, at 113 trees ha^{-1} and 50 trees ha^{-1} (error bars show the maximum and minimum values in each group)

4.4.3 Land equivalent ratios

The predicted land equivalent ratios for timber (including thinnings) and crop yield (assuming a full rotation) of the silvoarable systems at both 113 and 50 trees ha^{-1}, with a few exceptions, were between 1 and 1.4. Hence the Yield-SAFE model predicted that, under typical management, integrating crops and trees on the same land was more productive than growing them separately. The relationship between relative tree and crop yield suggested that the land equivalent ratio formed a convex arc with maximum values obtained when the trees and crops had similar relative yields and minimum values where either the tree or crop component was dominant (Figure 15). At each landscape test site, the land equivalent ratio at 113 trees ha^{-1} (Figure 15a) was greater than that at 50 trees ha^{-1} (Figure 15b), suggesting that in biophysical terms, 50 trees ha^{-1} was sub-optimal and more efficient use of resources in silvoarable systems could be achieved above this density.

The highest land equivalent ratios at both tree densities were associated with poplar, walnut and cherry systems in France (Figure 16a and Figure 16b). Oak and pine in Spain at both densities were associated with much lower land equivalent ratios. The reason for this is not clear; it may be that predicted growth of oak and pine was so slow that they were unable to make use of available resources at the densities used in the silvoarable systems. Alternatively, it may be that the crops competed more strongly for water than other trees of the same species. In either case, production benefits from oak and pine-based silvoarable systems in Spain appear to be limited unless tree densities can be increased without detriment to the relative yield of either component.

a) 113 trees per hectare **b) 50 trees per hectare**

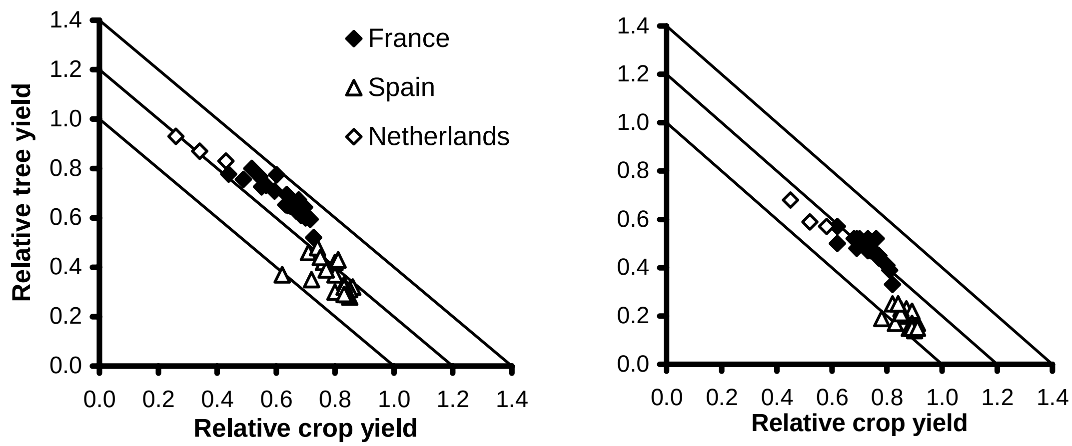


Figure 15 Predicted land equivalent ratio in France, Spain and the Netherlands

a) 113 trees per hectare **b) 50 trees per hectare**

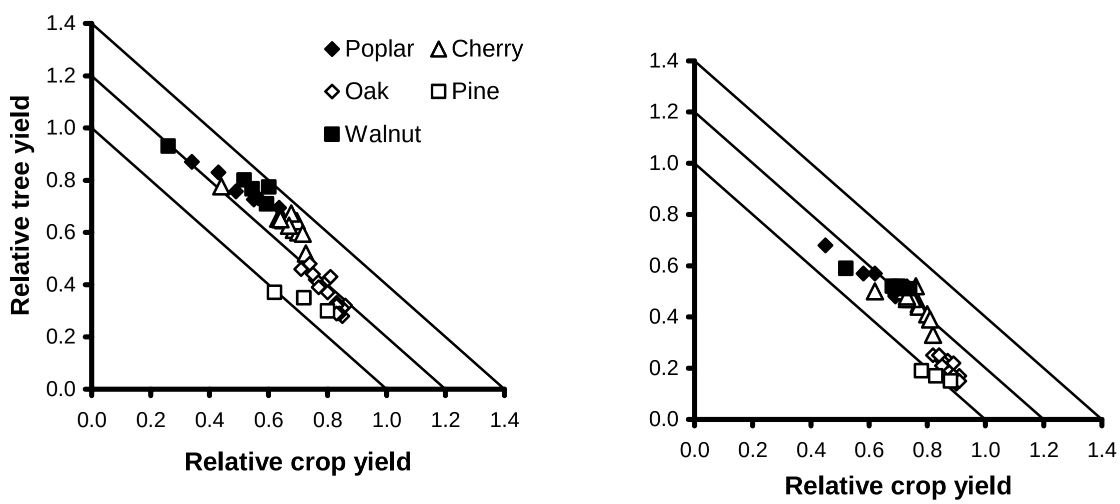


Figure 16: Predicted land equivalent ratios for poplar, cherry walnut, oak and pine

4.4.4 Plot-scale economic results

The annual time-series production data developed using Yield-SAFE and economic data for crop grants and crop revenue and costs, tree grants and tree revenue and costs for landscape test site were modelled in Farm-SAFE (Graves et al., 2005b). The economic performance of the arable, forestry and the silvoarable systems (113 trees ha^{-1} only) was compared using the equivalent annual value (*EAV*) (discount rate = 4%). The effects of zero grants, the pre-2005 grants and the post-2005 grants were also examined. As intercrop yields decreased over time due to tree growth, the crop rotation was optimised by ending intercrop production when the five-year moving-average of the intercrop net margin was zero.

4.4.4.1 Profitability with no grants

The equivalent annual values (at a discount rate of 4%) of the forestry systems with oak and stone pine in Spain, poplar and walnut in the Netherlands, and cherry in France were negative (Figure 17). Only walnut, due to the high value of the timber, and poplar, due to the short rotation, in France was profitable. The low profitability of forestry in the Netherlands was partly due to the opportunity cost of slurry manure management, as the application allowance was assumed to be zero for forest land. The equivalent annual values (4% discount rate) of the arable system were positive in Alcala la Real, Cardenosa El Espinar, Fontiveros, Olmedo and St Maria del Paramo in Spain, in Poitou Charentes and Centre in France and at all sites in the Netherlands, but negative in Torrijos, Ocaña and St Maria del Campo and at most sites in Franche Comté (i.e. at Dampierre and Vitrey). Positive values were associated with sites of high productivity. In Franche Comté, relatively high assignable fixed costs explained the negative values.

The equivalent annual values (4% discount rate) of the silvoarable systems in Spain were marginally below those for the arable system. By contrast, in France, values for the silvoarable systems with walnut, with poplar in Centre, and with wild cherry in Poitou Charentes and Franche Comté were higher than those for both arable agriculture and forestry. In the Netherlands, the values for the silvoarable system with poplar were marginally greater than the arable system, but the value for the silvoarable system with walnut was negative because of the long tree rotation and low value given to walnut timber.

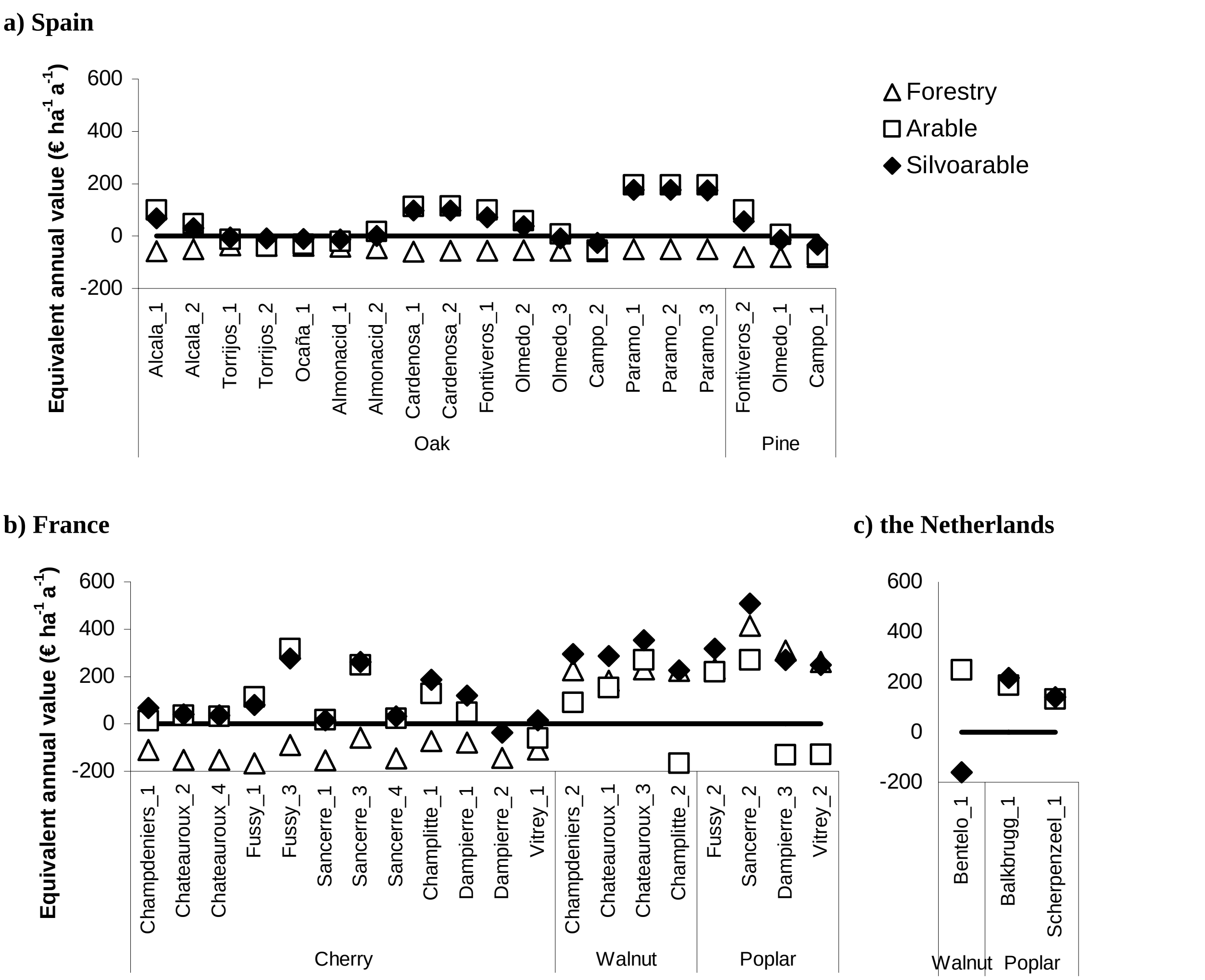


Figure 17: Equivalent annual value (discount rate of 4%) without grants of the arable, forestry and silvoarable (113 trees ha^{-1}) system in a) Spain, b) France and c) the Netherlands

4.4.4.2 <u>The long-term cash value of pre-2005 and post-2005 grant regimes</u>

Under the pre-2005 grant regime, the actual cash value (discount rate = 0%) of forest payments for the duration of the tree rotation was greatest in the Netherlands and lowest in France (Table 10). The assumed levels of arable compensation payments were marginally greater in the Netherlands than in France, and both were much greater than in Spain.

Within Spain, support for silvoarable agroforestry was lower than for forestry and the arable system because of ineligibility for tree grants and reduction of the arable compensation payments by twice the proportion of the canopy area of the trees. In France, in Poitou Charentes and Centre, arable payments were at least five-times the value of forestry payments and the value of silvoarable payments was marginally less than that for arable systems. In Champlitte, Dampierre and Vitrey in Franche Comté, there were no forestry payments and hence the greatest level of support was for arable systems. For poplar sites, payments for all systems were relatively low because of the 20- rather than the 60-year rotation. Support for walnut and poplar forestry in the Netherlands was identical because they were both temporary, production-based systems. Since arable payments were dependent on the length of the tree rotation, they were greater for walnut (Bentelo) rather than for poplar (Balkbrug and Scherpenzeel). In each case, the support for silvoarable systems was less than for forestry and arable systems, as no payments were received for the tree component.

The actual cash value of each system in the post-2005 payment scenario and the change, relative to the pre-2005 scenario was determined (Table 10). The greatest relative change was predicted for Spain, where forestry payments were greatly reduced, due to compensation being limited to 10 years, while for arable and particularly for silvoarable systems, payments were predicted to increase. The predicted value of the new single farm payment at Alcala la Real, and St Maria del Paramo and St Maria del Campo was greater than pre-2005 area payments, as support for non-arable activities on typical farms in these areas was assumed to be re-allocated on an area basis. The large relative increase of the cash value of payments in the silvoarable systems demonstrated the disadvantage of the system under pre-2005 regime. In France, there was no change for forestry, and only marginal changes for arable systems due to modulation under the single farm payment. For silvoarable systems, scenario 1 was similar to the pre-2005 regime but marginal benefits were evident under scenario 2. In the Netherlands, the major change was due to the reduction in the compensation payments associated with forestry from 18 to 10 years.

Table 10: The predicted value of government support (€ ha^{-1}), over a full tree-rotation, for forestry, arable and silvoarable systems in the pre-2005 grant regime, and the predicted change in that support in a post-2005 grant regime (scenario 1 and scenario 2)

	Pre-2005 government support				Predicted net change in support with the post-2005 grant regime			
Land unit	Rotation (a)	Forestry	Arable	Silvoarable	Forestry	Arable	Silvoarable scenario 1	Silvoarable scenario 2
Spain								
Alcala 1	60	6860	5170	2010	**-2940**	8030	10010	11408
Alcala 2	60	6860	5170	2690	**-2940**	8030	9320	10728
Torrijos 1	60	9380	3870	1410	**-4190**	210	820	1256
Torrijos 2	60	9380	5170	1920	**-4180**	270	1790	2378
Ocaña 1	60	9380	5170	1770	**-4190**	350	2120	2864
Almonacid 1	60	9380	3870	1380	**-4190**	600	2010	2712
Almonacid 2	60	9370	8770	4080	**-4180**	**-1030**	2980	3886
Cardenosa 1	60	8860	5810	2900	**-3380**	**-590**	1850	2538
Cardenosa 2	60	8860	5810	2670	**-3390**	**-590**	2080	2768
Fontiveros 1	60	8850	6200	2940	**-3380**	950	3570	4430
Olmedo 2	60	8860	5160	2260	**-3390**	600	2990	3718
Olmedo 3	60	8860	6100	2520	**-3380**	**-340**	2720	3458
Campo 2	60	8860	6460	2610	**-3380**	1990	4160	6058
Paramo 1	60	8860	6760	3080	**-3390**	2500	5350	6402
Paramo 2	60	8860	6760	3080	**-3390**	2500	5350	6402
Paramo 3	60	8860	6760	3060	**-3390**	2500	5370	6422
Fontiveros 2	60	8000	6200	2060	**-2960**	950	4450	5335
Olmedo 1	60	8010	6100	1780	**-2970**	**-340**	3470	4223
Campo 1	60	8010	5810	1050	**-2970**	1790	2640	4263

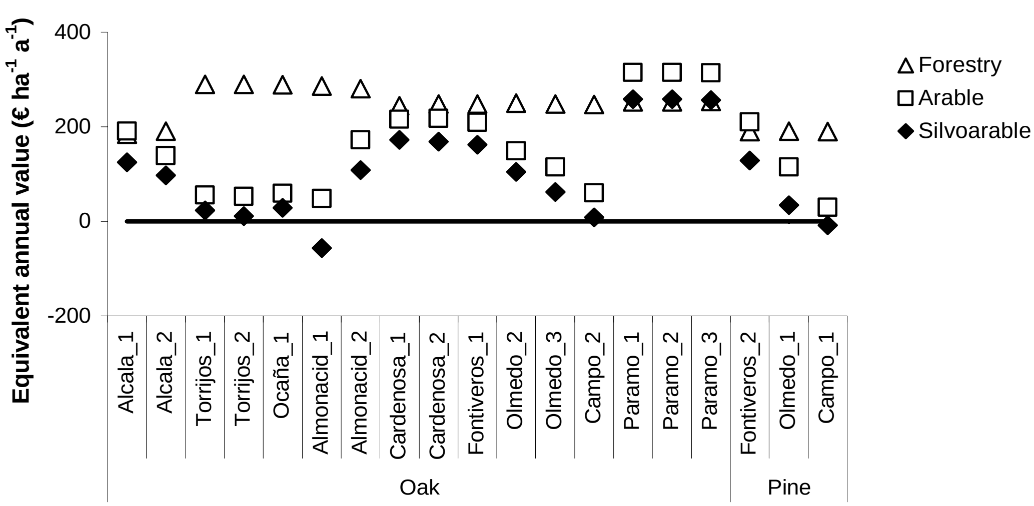
Table 10 (continued)

Land unit	Rotation (a)	Pre-2005 government support: Forestry	Arable	Silvoarable	Predicted net change in support with the post-2005 grant regime: Forestry	Arable	Silvoarable scenario 1	Silvoarable scenario 2
France								
Champdeniers 1	60	4440	21180	16130	0	0	**-390**	1564
Fussy 3	60	3840	21090	19590	0	**-30**	**-430**	1865
Sancerre 3	60	3840	20860	19380	0	**-70**	**-460**	1805
Fussy 1	60	3840	21090	19590	0	**-30**	**-420**	1865
Chateauroux 2	60	3840	21000	19510	0	**-50**	**-440**	1835
Chateauroux 4	60	3840	21000	19510	0	**-50**	**-440**	1835
Sancerre 4	60	3850	20940	19450	0	**-150**	**-530**	1735
Sancerre 1	60	3840	20860	19380	0	**-70**	**-460**	1805
Champlitte 1	60	0	20080	11880	0	**-100**	**-60**	2169
Dampierre 1	60	0	21040	15320	0	**-1300**	**-1550**	868
Vitrey 1	60	0	19840	14450	0	**-60**	**-50**	2759
Dampierre 2	60	0	20940	12700	0	**-1200**	**-730**	856
Champdeniers 2	60	4270	21180	16440	0	0	**-700**	1567
Chateauroux 3	60	3670	20800	19630	0	150	**-570**	2027
Chateauroux 1	60	3680	21000	19820	0	**-50**	**-750**	1837
Champlitte 2	60	0	19880	3920	0	100	20	3449
Sancerre 2	20	2720	6940	6850	0	**-10**	**-540**	610
Fussy 2	20	2720	6960	6870	0	60	**-480**	680
Dampierre 3	20	0	7080	3870	0	**-500**	**-280**	609
Vitrey 2	20	0	6600	3610	0	**-10**	**-10**	877
The Netherlands								
Bentelo	60	11810	23000	5230	**-1980**	**-1820**	**-410**	2811
Balkbrugg	20	11810	8000	3640	**-3310**	0	0	1026
Sherpenzeel	20	11810	8000	4370	**-3640**	0	0	1096

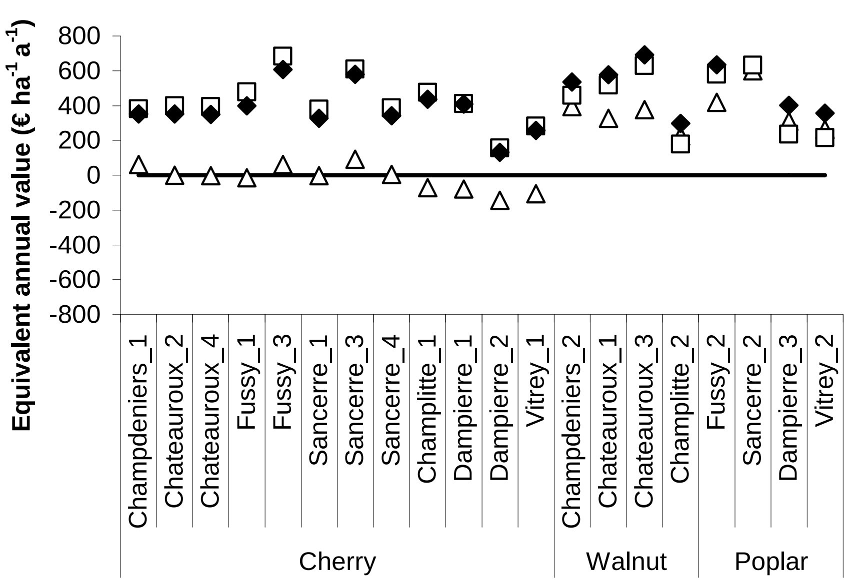
4.4.4.3 Equivalent annual value under pre-2005 grant regime

In the pre-2005 grant regime (Figure 18), the equivalent annual values (4% discount rate) of forestry in Spain was generally higher than those for arable systems, except where crop yields were high. Because of the low level of government support, the equivalent annual value of the silvoarable systems was generally lower than for forestry and arable systems. In France, the equivalent annual value for the arable systems, tended to be greater than that for silvoarable agroforestry with wild cherry, which was much greater than that for wild cherry forestry. Hence, silvoarable agroforestry offered the most profitable means of establishing cherry trees at these sites. The predicted equivalent annual values of the silvoarable systems with poplar and walnut systems in France were higher than that for both forestry and arable systems. In the Netherlands, the conventional arable systems were the most profitable, followed by silvoarable agroforestry. Thus in the Netherlands, silvoarable systems also appeared to provide a more profitable means of establishing trees in the landscape.

a) Spain



b) France



c) the Netherlands

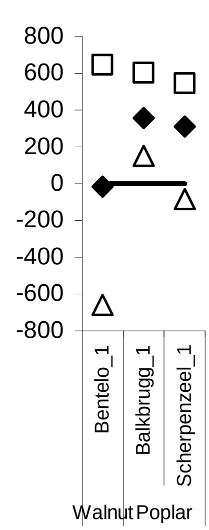


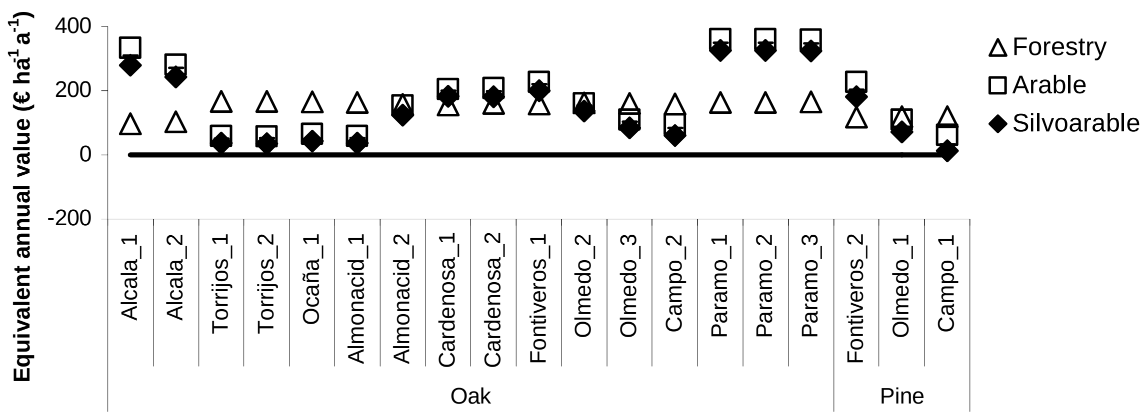
Figure 18: Equivalent annual value (4% discount rate) of the arable, forestry and silvoarable (113 trees ha^{-1}) system in a) Spain and b) France and c) the Netherlands, assuming the pre-2005 grant regime.

4.4.4.4 Equivalent annual value under post-2005 grant regime

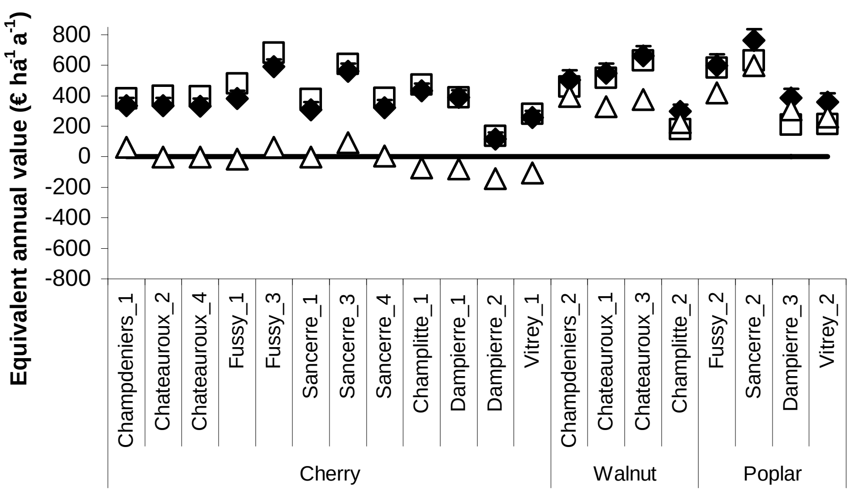
In the post-2005 (Figure 19), compared to the pre-2005 (Figure 18), grant regime in Spain, the equivalent annual value of forestry was predicted to be reduced whilst it was predicted to increase for arable and silvoarable systems despite modulation (Figure 18). In France, the values for the equivalent annual value were generally similar to those under the pre-2005 regime. For silvoarable systems, the pessimistic scenario, scenario 1, resulted in marginal reductions, while the optimistic scenario, scenario 2, resulted in marginal increases. In the Netherlands, a substantial decrease in the equivalent annual value of forestry was predicted, whilst the change in the equivalent annual value of arable systems was marginal. For silvoarable agroforestry, little change was predicted for scenario 1, but a small and consistent increase was predicted for scenario 2.

The net effect of the above changes was most significant in Spain. Under the pre-2005 scenario, forestry systems were consistently more profitable than silvoarable systems. Under the post-2005 scenario, silvoarable agroforestry was predicted to be more profitable than forestry in almost 50% of cases, although both systems were predicted to remain less profitable than arable agriculture. At sites in France and the Netherlands, the ranking of the systems in the post-2005 and pre-2005 regimes were similar.

a) Spain



b) France



c) the Netherlands

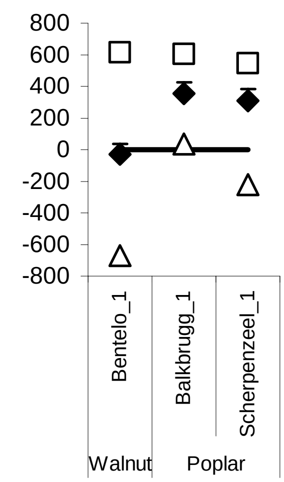


Figure 19: Equivalent annual value (4% discount rate) of a forestry, arable, and silvoarable (113 tree ha^{-1}) system in a) Spain and b) France and c) the Netherlands, assuming the 2005 grant scenario 1 (error bars show the equivalent annual value for scenario 2)

4.4.5 Farm-scale feasibility

Under the pre-2005 grant-regime in Spain, it was not profitable to re-plant arable land with a silvoarable system. This was due to low timber volume and value, the lack of tree grants and the loss of arable area payments by twice the canopy area of the tree component. By contrast, establishing forestry on arable land was predicted generally to increase farm profitability

(Figure 20). In France, establishing silvoarable agroforestry was predicted to increase farm profitability when it involved walnut or poplar, and decrease it if it included wild cherry. In each case, silvoarable systems improved farm profitability relative to forestry on the same area of land. In the Netherlands, both forestry and silvoarable systems reduced farm profitability.

Under the post-2005 grant regime in Spain, replanting arable land with silvoarable systems continued to result in reduced farm profitability. Replanting arable land with forestry was predicted to increase farm profitability at five sites (Torrijos, Ocaña, Almonacid de Zorita, Olmedo and St Maria del Campo), and decrease it at the other four. In France, farm profitability increased following the establishment of silvoarable systems with walnut and poplar, and decreased with silvoarable systems using wild cherry; in both cases silvoarable systems were more profitable than forestry. In the Netherlands, there was no advantage to introducing silvoarable systems or forestry in comparison with the status quo.

An analysis of the frequency with which silvoarable systems increased profitability relative to the status quo (Figure 20) showed that in Spain there were no cases where farm profitability was improved by establishing silvoarable systems. Instead government support favoured the establishment of forestry, and this was attractive in about 80% of cases under the pre-2005 grant regime. The post-2005 regime was predicted to reduce the relative profitability of forestry, but forestry still remained financially attractive on about 50% of the selected farms. In France, under the pre-2005 grant regime, silvoarable systems were predicted to increase farm profitability in approximately 50% of cases. This frequency remained similar under scenario 1 of the post-2005 grant regime, and increased to 80% under scenario 2. The proportion of farms where forestry was attractive (20%) was less than for silvoarable systems and was the same for both the pre-2005 or post 2005 regimes. In the Netherlands (not shown), the introduction of forestry and silvoarable systems always reduced farm profitability, under the pre-2005 and post-2005 payment scenarios.

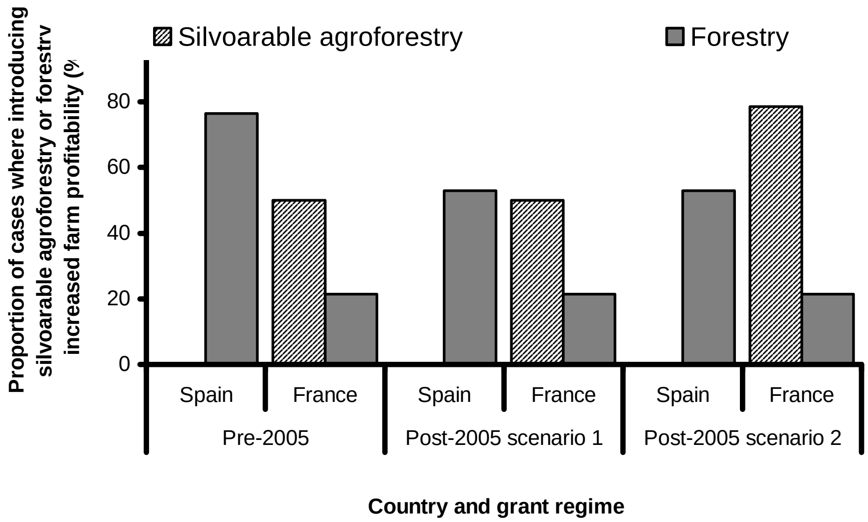


Figure 20: Proportion of farms where the farm net present value was improved compared with the status quo by the introduction of silvoarable systems or forestry (Spain: n =17; France: n = 14)

In Spain the use of silvoarable systems was preferable to forestry in 12% of cases under the pre-2005 grant regime and 50% of cases in scenarios 1 and 2 of the post-2005 grant regime. In France and the Netherlands, farm profitability was always increased with the use of silvoarable rather than forestry systems (Figure 21). Hence, in Spain, forestry generally provided the most cost effective method of establishing trees under the pre-2005 regime, an advantage predicted to disappear under post-2005 regime. In France and the Netherlands, silvoarable systems with walnut, wild cherry, and poplar provided the most profitable means of establishing trees on farms irrespective of grant regime.

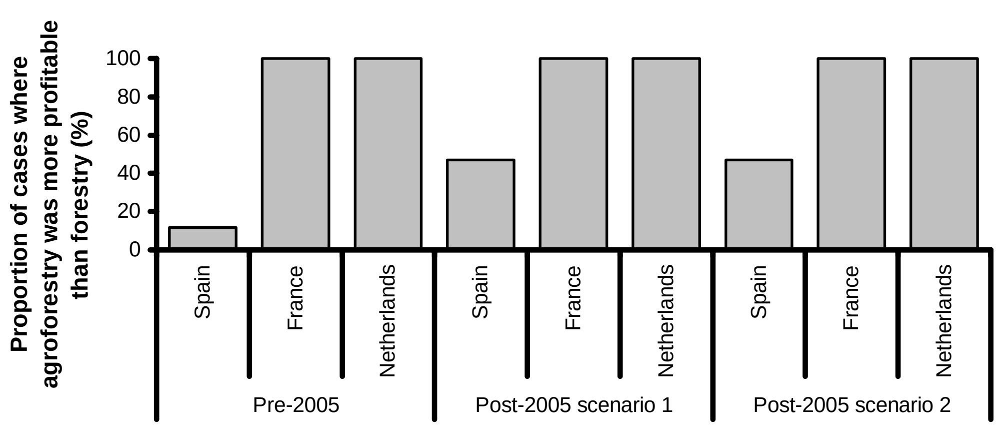


Figure 21: Frequency with which silvoarable systems outperformed forestry (Spain: n = 17; France: n = 14; the Netherlands: n = 3)

4.5 Summary and recommendations

Using a geographical information system, a statistical analysis of climatic, topographic and land classification data was used to select 19 landscape test sites in Spain, France and the Netherlands. Within each site, land use, soil depth and texture, and elevation were digitised. Daily weather data were generated for each site using a weather generator. Proportional differences in solar radiation and soil water holding capacity were calculated and used in cluster analysis to divide the arable land at each site into between one and four land units. A biophysical model called "Yield-SAFE" was developed and calibrated for potential yields of a range of tree and crop species. Typical forestry and arable systems and associated management regimes were determined for each land unit and Yield-SAFE was calibrated for actual tree and crop yields at each site. The calibrated model was then used to calculate daily values of tree and crop yields for a forestry, arable and agroforestry system at each land unit according to changes in solar radiation, soil depth and texture. Financial data for forestry, arable, and silvoarable production at each site were collected and four grant scenarios were described (no grants, a pre-2005 scenario, and two possible post-2005 scenarios). The financial data was combined with the physical values in an economic model called "Farm-SAFE", and the equivalent annual value (discount rate = 4%) at a plot-scale and the infinite net present value at a farm-scale were used to examine the profitability of different systems.

The Yield-SAFE biophysical model predicted lower timber yields and crop yields per hectare for silvoarable systems compared to the forestry and arable systems respectively (Figure 13). However, the total productivity of the silvoarable system, as determined by a land equivalent ratio, was predicted to be between 100 and 140% of that for the monoculture systems (Figure 15 and Figure 16). High land equivalent ratios were achieved with a tree stand density of 113 rather than 50 trees ha^{-1}, suggesting that the high density system made fuller use of the available light and water resources. The highest ratios were obtained by integrating deciduous trees and autumn-planted crops, which were complementary in terms of light use (Figure 14). The lowest ratios were obtained from evergreen tree species in Spain, where productivity appeared to be constrained by the slow growth of the trees and low soil water availability (Figure 14).

At a plot scale, the economic performance of the systems was compared in a zero grant scenario (Figure 17). In Spain, arable systems were marginally more profitable than silvoarable systems with oak or stone pine, which in turn were more profitable than forestry systems with the same species. By contrast in France, silvoarable systems with walnut in each of three regions, poplar in one region, and wild cherry in two regions were more profitable than arable and forestry systems. In the Netherlands, silvoarable systems with poplar, but not walnut, were predicted to be more profitable than the described arable system. However, both the poplar and walnut silvoarable systems were more profitable than forestry.

Under pre-2005 grants (Figure 18), support for silvoarable systems in Spain and the Netherlands was substantially lower than for arable and forestry systems. Hence, the profitability of silvoarable systems was always less than for arable or forestry systems. In France, support for silvoarable systems was marginally lower than for arable systems but significantly higher than for forestry systems. Hence it was predicted that silvoarable systems with poplar and walnut could be more profitable (at a 4% discount rate) than both forestry and arable systems. Silvoarable systems with cherry although more profitable than forestry were predicted to be less profitable than arable systems. In the Netherlands, silvoarable systems were more profitable than forestry, but less profitable than arable systems.

Under two possible post-2005 grant regime (Figure 19), the relative value of support for forestry in Spain was predicted to decrease, whilst for silvoarable and arable systems it was predicted to increase. In France and the Netherlands the relative value of support for silvoarable systems compared to arable and forestry systems remained similar to the pre-2005 regime for scenario 1, and increased marginally for scenario 2. Hence the profitability of silvoarable systems in Spain increased and frequently exceeded the profitability of forestry systems, but remained marginally less profitable than arable systems. In France and the Netherlands, little relative change in profitability between the systems was predicted.

At a farm-scale and under both pre-2005 and post-2005 grants in France (Figure 20), planting arable land with silvoarable systems of walnut and poplar increased farm profitability, while silvoarable systems with cherry reduced farm profitability. In Spain and the Netherlands, silvoarable systems consistently reduced farm profitability in comparison with the arable status quo. However, in both France and the Netherlands, silvoarable systems were a more cost-effective way of establishing trees on the farm than forestry (Figure 21). In Spain, under pre-2005 grants, silvoarable systems were a less cost-effective means of establishing trees than forestry. However, under post-2005 grants, silvoarable systems were predicted to be a most profitable means of establishing trees in half the examined cases.

A number of recommendations regarding further research can be made. Predictions are subject to uncertainty and this could be examined using sensitivity analysis or stochastic modelling. Certain baseline data could also be re-examined. The recorded value of walnut timber in the Netherlands and France differed greatly, even though both countries are part of a free-trade zone. This strongly influenced the relative profitability of walnut systems in these countries. The assumption regarding prohibition of slurry manure application in the Netherlands in forests also had an important effect. If this is a true opportunity cost, the establishment of productive forests on farms is unlikely to be attractive, unless the opportunity cost is removed or payment schemes can account for it. Assumptions regarding beating-up, tree management and the extent of payments could also be re-assessed for Spain. Tree mortality is likely to be high due to difficult conditions and should be accounted for; the assumptions regarding pruning and thinning costs in Spain may be valid for traditional management of widely spaced trees in open woodlands (dehesas), but invalid for forestry and silvoarable systems, even if these are established within areas where dehesas predominate. Finally, the assumptions and value of post-2005 grants should be re-assessed when the changes are implemented.

4.6 Conclusion

The process used to model plot- and farm-scale economics of arable, silvoarable and forestry systems in three European countries has been described. This integrated the use of geographical information systems with a biophysical model of tree, crop and integrated tree and crop growth, and an economic model developed during the SAFE project.

Under the economic conditions envisaged in the analysis, the most financially attractive silvoarable systems tended to have a land equivalent ratio that was substantially above one. Conditions that most favoured a high land equivalent ratio appeared to be the use of relatively high tree-densities to make full use of available resources, the use of deciduous trees and autumn-planted crops to make complementary use of light, and a high soil water availability to ensure that extra biomass production could be sustained. Conversely, it appeared that low ratios were associated with low tree density, evergreen trees, spring-planted crops, and low soil water availability.

Silvoarable agroforestry was most financially attractive where both components of the system were profitable as a monoculture since an unprofitable or relatively unprofitable component tended to reduce the profitability of the mixed system. In addition, the profitability of silvoarable agroforestry tended to be maximised if the profitability of the forestry and agricultural system were similar. Under the two proposed post-2005 grant regimes, it is predicted that silvoarable systems with walnut and poplar in France could provide a profitable alternative to arable or forestry systems. In Spain, it appeared that holm oak and stone pine could be integrated into arable systems without significantly reducing arable production for many years. Since these trees are of ecological and landscape importance, rather than productive importance, additional support in the form of an agri-environment payment would be justified. A moderate annual amount would be sufficient to overcome income losses caused by yield reductions and encourage establishment for non-productive benefits. In the Netherlands, the low value of timber and an assumed opportunity cost of losing arable land for slurry manure application made silvoarable and forestry systems relatively unattractive compared with arable systems.

Chapter 5

Integrating environmental and economic performance to assess modern silvoarable agroforestry in Europe

Based on Palma J, Graves A, Burgess P, Herzog F, 2006, Integrating profitability and environmental indicators to assess modern silvoarable agroforestry in Europe, Ecological Economics, submitted

5.1 Abstract

The environmental and economic performance of silvoarable agroforestry in Europe is highly variable. Multi-criteria analysis, using the PROMETHEE outranking approach, was used to evaluate the integrated performance of silvoarable agroforestry relative to a status quo, on hypothetical farms in nineteen landscape test sites in Spain, France, and the Netherlands. The silvoarable scenarios allocated a proportion of the hypothetical farms (10 or 50%) to silvoarable agroforestry at two different tree densities (50 or 113 trees ha^{-1}) on two different qualities of land (best or worst quality land). The status quo (conventional arable farming) was also assessed for comparison. The criteria used in the evaluation (soil erosion, nitrogen leaching, carbon sequestration, landscape biodiversity, and infinite net present value) were assessed at each landscape test site; infinite net present value was assessed under six levels of government support. In France, the analysis showed, assuming equal weighting between environmental and economic performance, that silvoarable agroforestry was preferable to conventional arable farming. The best results were observed when agroforestry was implemented on 50% of the highest quality land on the farm; the effect of tree density (50-113 trees ha^{-1}) was small. By contrast, in Spain and the Netherlands, the consistently greater profitability of conventional arable agriculture relative to the agroforestry alternatives made overall performance of agroforestry systems dependent on the proportion of the farm planted, and the tree density and land quality used.

5.2 Introduction

The environmental and economic performance of silvoarable agroforestry (SAF) in Europe is highly variable (Chapters 3 and 4). This variability results from the interaction of many factors influencing outputs of SAF. For example, different economic and environmental results are obtained in different European regions due to the many combinations of biophysical and management conditions, such as choice of tree and crop species, national legislation, market conditions, and regional policies.

Within the context of agricultural policy which takes into consideration environmental performance (Pezaros, 2001, van Dijk, 2001), an analysis of land-use systems should consider both economic and environmental impacts. However, environmental and economic evaluations are complex and tend to be undertaken separately and drawing together these separate analyses becomes impossible, because of the different assumptions and scenarios used. For example, environmental results for agroforestry systems obtained by Udawatta *et al.* (2002) or Nair & Graetz (2004) cannot be linked to economic results obtained in different contexts (e.g. Thomas, 1991, Thomas & Willis, 1997). Whilst there have been some studies on the economic performance of agroforestry (Willis *et al.*, 1993, Dupraz *et al.*, 1995, Mercer *et al.*, 1998, Requillart *et al.*, 2003, Montambault & Alavalapati, 2005), environmental assessments have been sparse (e.g. Burgess, 1999, Stamps *et al.*, 2002, Montagnini & Nair, 2004, Thevathasan & Gordon, 2004, Klaa *et al.*, 2005) and no integrated assessments have been conducted to date.

Integrated assessments require the comparison of independent indicators (e.g. soil erosion, nitrogen leaching, profitability) with different physical units (e.g. t ha^{-1} a^{-1} of soil loss, kg ha^{-1} a^{-1} of leached nitrogen, € ha^{-1} a^{-1} of profit). One way of integrating the results of such indicators is to monetarise them and computes an overall profitability, which therefore includes the value of the environmental results. The advantage of this approach lies in the ease of communication of the final result – an integrated profitability. Also, monetarising environmental costs and benefits emphasizes their value and importance to individuals and society. However, there are a multitude of difficulties related to monetarisation. Whilst estimates of the economic value of soil loss can be based on impacts on crop yields, or the cost of nitrogen leaching derived from the cost of water purification, the economic value of landscape biodiversity for example is more difficult to assess. Although economic values can be obtained through various methods, such as contingency valuation, the validity of results obtained from such approaches has been questioned (Mitchel & Carson, 1989, Hanley & Spash, 1993, Pethig, 1993). Moreover, an integrated assessment based on monetarisation is not always transparent. Whilst the financial benefit of agricultural production goes to the farmer, environmental benefits (or costs) may be relevant to either the farmer (e.g. soil erosion, which reduces profitability), both the farmer and society (e.g. nitrogen leaching which creates additional fertiliser costs for the farmer and water purification costs for society), or society as a whole (e.g. landscape biodiversity). These distinctions are lost when integrated in a single integrated monetary value.

Hence, multi-criteria decision analysis (MCDA) was used instead, since this could be used to evaluate the relative importance of the selected criteria and reflect their importance in the final result (Belton & Stewart, 2002). Environmental results (Chapter 3) and economic results

(Chapter 4) obtained previously during the European Union "Silvoarable Agroforestry for Europe" project (Dupraz *et al.*, 2005) were assessed using MCDA to provide an integrated analysis of the environmental and economic benefits of SAF. This chapter therefore provides an integrated overview of the impact of SAF systems which were considered to be suitable for Europe (EC, 2005a, Lawson *et al.*, 2005, Reisner *et al.*, 2006).

5.3 Material and methods

5.3.1 Environmental and economic data

An environmental classification of Europe, derived from a statistical analysis of climatic and topographic data (Metzger *et al.*, 2005), was used to randomly select 19 landscape test sites (LTS) of 4 km x 4 km in the dominant environmental classes of Spain, France and the Netherlands (Figure 9, page 36). In each LTS, the environmental assessment comprised analysis of soil erosion, nitrate leaching, carbon sequestration and landscape biodiversity (Chapter 3). In Chapter 4 was partially shown profitability in terms of the infinite net present value (iNPV) for two extreme payment scenarios (Table 7, page 66). However the full assessment was made to a total of six levels of government support of SAF. The first scenario considered no support at all (iNPV_0), the second considered the support received from the European Union's Common Agricultural Policy (CAP) up to 2004 (iNPV_04) and four schemes considered future options based on CAP reform in the Rural Development Regulation (EC, 2005a) and included use of single farm payments (SFP) (EC, 2004a). The first of these considered that SFP would be based on the percentage of crop area in silvoarable system (iNPV_05_1.1), the second assumed SFP for the whole area (iNPV_05_1.2), the third (iNPV_05_2.1) considered SFP as for NPV_05_1.1, but included additional tree payments as outlined in the Rural Development Regulation, and the fourth (iNPV_05_2.2) considered SFP as for iNPV_05_1.2, but with the additional tree payments. The tree payments were considered to be equivalent to half of the costs of tree establishment during the first four years of the tree rotation (*see sections 4.3.5.3 4.3.5.4*, page 64). In order to harmonize with MCDA terminology, the "assessments" and "land-use scenarios" will be called *criteria* and *alternatives*, respectively.

In each LTS, the environmental and economic criteria for SAF were modeled for different alternatives of tree density (50 or 113 trees ha^{-1}), land quality (best or worst) and share of farmland converted to SAF (10 or 50%). Tree lines were assumed to be planted along contour lines, which is important for erosion control (see *section 3.5.1*, page 44). A total of eight alternatives were compared with conventional arable land-use (status quo) under six different payment schemes (Figure 22).

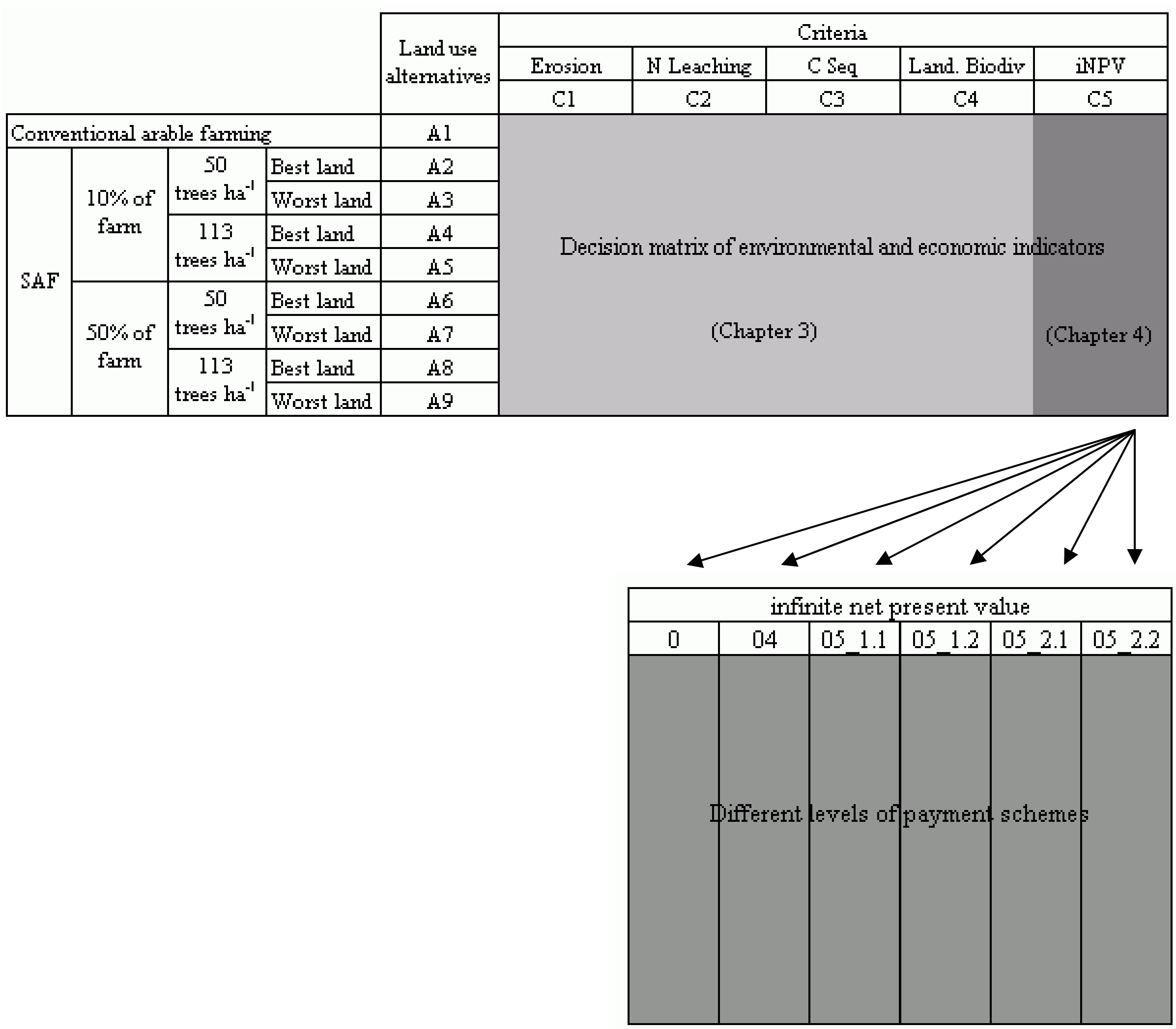


Figure 22: Definition of the alternatives and design of the decision matrix with floating levels of payments for the economic criteria. SAF, Silvoarable Agroforestry; C Seq, Carbon Sequestration; Land. Biodiv., Landscape Biodiversity; iNPV, infinite net present value.

5.3.2 MCDA outranking approach

MCDA outranking methods focus on pair-wise comparisons of alternatives where the starting point is a decision matrix describing the performance of the alternatives to be evaluated with respect to identified criteria (Belton & Stewart, 2002). The PROMETHEE II method (Brans

& Vincke, 1985, Brans *et al.*, 1986, Brans & Mareschal, 1990, Brans & Mareshal, 2002) was used as it enables the complete pre-order of alternatives, facilitating the tracing of the final performance rank..

A general characteristic of PROMETHEE and other outranking methods is that all k alternatives are compared in a pair-wise manner, separately for each criterion. To formalize this, let $g_j(L_a)$ and $g_j(L_b)$ be the values of two land use alternatives L_a and L_b for criterion C_j. The difference between the two indicator values is denoted as $d_j(L_a, L_b) = g_j(L_a) - g_j(L_b)$, which measures the extent to which L_a 'outperforms' L_b in criterion C_j. The "preference function" $\Pi_j(L_a, L_b)$ maps this difference into a preference score which is between 0 and 1 and mirrors how strongly L_a is preferred to L_b in terms of C_j. We opted for the "Type II – quasi-criterion function" with zero as the reference parameter: if L_a outperforms L_b, i.e. if $d_j(L_a, L_b) > 0$, then the preference of L_a to L_b is $\Pi_j(L_a, L_b) = 1$. If L_a and L_b perform equally or if L_b outperforms L_a, i.e. if $d_j(L_a, L_b) \leq 0$, then the preference of L_a to L_b is $\Pi_j(L_a, L_b) = 0$.

The preference was calculated separately for each criterion and for all pairs of land use alternatives. These preferences were aggregated over the criteria to obtain a total preference for each pair of land use alternatives. The "total preference" of L_a to L_b was then calculated as the weighted sum of the pairwise preferences for all criterion indicators (j = 1 to 5), the latter associated to a certain weight of importance (w):

$$\Pi(L_a, L_b) = \sum_{j=1}^{5} w_j \Pi_j(L_a, L_b) \quad \textit{with} \quad \sum_{j=1}^{5} w_j = 1 \qquad \textit{(Equation 18)}$$

In this primary evaluation, the weights (w) were given according to a neutral preference (*see section 5.3.4 below*).

Equation 18 reflects only two land use alternatives (L_a, L_b). In the case of this evaluation there were nine alternatives. Therefore the total preference was calculated for all pairs of land use alternatives (L_u, L_v) (u,v = 1 to 9). The nine alternatives were ranked in two different ways. The first reflects how strongly an alternative L_u dominates all the other alternatives L_v (v = 1 to 9) – $\Phi^+(L_u)$ in Equation 19 - and the second reflects how strongly L_u is dominated by all the other alternatives L_v (v = 1 to 9) – $\Phi^-(L_u)$ in Equation 20.

$$\Phi^+(L_u) = \sum_{v=1}^{9} \Pi(L_u, L_v) \qquad \textit{(Equation 19)}$$

$$\Phi^-(L_u) = \sum_{v=1}^{9} \Pi(L_v, L_u) \qquad \textit{(Equation 20)}$$

Finally, a performance rank Φ (L_u) considering all alternatives was computed for each alternative:

$$\Phi(L_u) = \Phi^+(L_u) - \Phi^-(L_u) \quad \textit{with} \quad u = 1,\ldots,9 \qquad \textit{(Equation 21)}$$

Each land use alternative was then ranked according to the integrated environmental and economic performance of each alternative. The higher the performance rank, the higher the preference of the alternative.

5.3.3 MCDA design

The MCDA had nine alternatives and five criteria to evaluate (Figure 22). For each of the nineteen LTS, a decision matrix was built (Figure 22) with all the criteria assessed in Chapters 3 and 4. The erosion and nitrogen leaching assessment values were recalculated in order to show reduction of erosion and nitrogen leaching comparatively to the status quo, so higher values could correspond numerically to “better alternatives”.

In Chapter 4, six different economic payment schemes under different subsidy levels were modeled. Therefore six decision matrices were built for each LTS keeping constant the environmental indicators and varying the economic indicator (Figure 22). A total of 114 decision matrices were analysed.

In each LTS, the alternatives were ranked with Φ (Equation 21) and a representative average of the performance of each alternative was calculated for all 19 LTS. A subgroup average at country level was also calculated as the integrated economic and environmental results varied substantially within each country (see Chapters 3 and 4).

5.3.4 Weighting criteria

Different stakeholders can have different objectives depending on their personal values and on their socio-economic circumstances. As we did not wish to adopt the position of a specific stakeholder (e.g. an NGO might rate environmental criteria higher than profitability, whereas farmers might rate profitability higher than environmental criteria), for this MCDA a neutral weight distribution was used. Thus, the environmental and economic criteria were considered as two groups with the same weight (0.5 each). Because the environmental assessment involved four criteria, each one had the value of (0.5/4). In other words, the sum of all the environmental outputs of each land use alternative was tested against its economic performance.

For a better starting point in the evaluation, we also ran the analysis in a “uni-criterion” analysis mode to have an overview of the performance of each alternative under each criterion independently. This was done using Equation 18, by setting the weight of the selected criterion to 1 and the weights of all other criteria to 0.

5.4 Results and discussion

5.4.1 "Uni-criterion" analysis

The performance of each alternative was evaluated for each criterion independently to improve understanding of how each alternative was influenced by individual criterion. The results are shown on a per country basis (Figure 23).

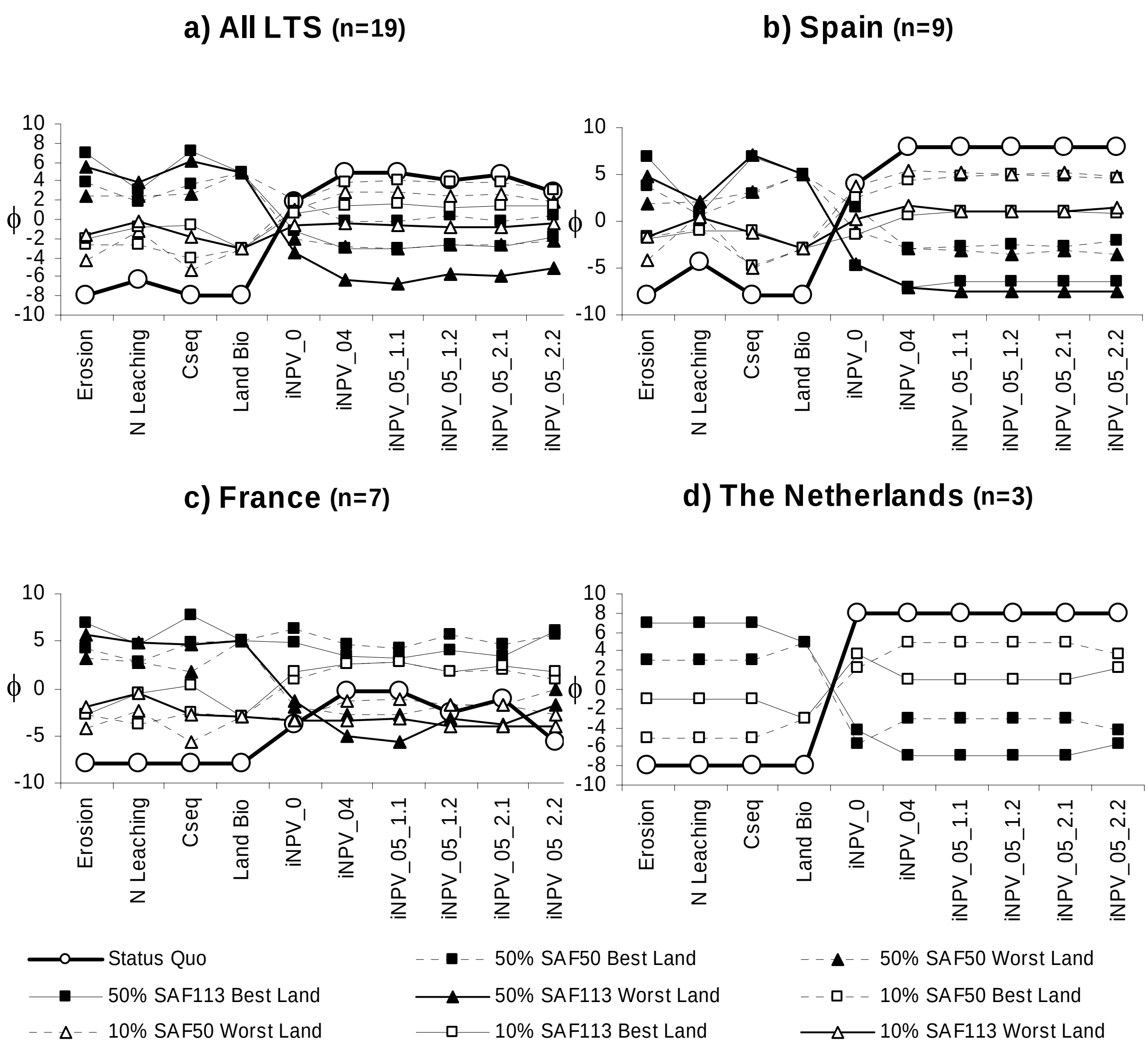


Figure 23: Preference rank of each alternative for each criterion for a) all landscape test sites (LTS) and for b) Spain, c) France and d) The Netherlands separately. Reduction of erosion and N leaching are considered. Note: in The Netherlands there was only one homogeneous landscape and therefore only one quality of land was assessed.

The difference in effect between environmental and economic criterion are clearly visible in Figure 23. The aggregate result for all 19 LTS showed that alternatives with the best environmental performance were those with the worst economic performance (Figure 23a) except for iNPV_0. Under this "zero subsidy" assumption, the alternative of implementing SAF with 50 trees ha^{-1} on 10 or 50% of the farm showed similar or slightly higher preference than the status quo alternative. Under all levels of government support however, the status quo alternative was preferable to SAF. But SAF was most preferable when soil erosion, nitrogen leaching, carbon sequestration and landscape biodiversity were evaluated. The best environmental and worst economic evaluations were associated with SAF implemented on 50% of the farm (filled symbols).

These patterns were also observed in Spain and the Netherlands (Figure 23b, d). In France however, the SAF alternatives on high quality land were preferable to the status quo alternative for all environmental and economic criteria, regardless of the tree density or the proportion of the farm to be converted (Figure 23c). This was in part due to relatively high financial returns from the timber produced in the SAF systems, since the trees selected were valuable (walnut) or produced rapid returns (poplar (*Populus* sp.)) (Chapter 4).

5.4.2 Multicriteria analysis

In the next step, the environmental and economic criteria were analyzed together. The aggregate results for all 19 LTS showed that the SAF alternatives were preferable to the status quo alternative under all payment schemes, especially when there was no government support (Figure 24a – iNPV_0). However, this aggregate response for all three countries masks important differences between the countries that occurred as a result of regional variations in biophysical conditions, selected tree and crop species, and market dynamics.

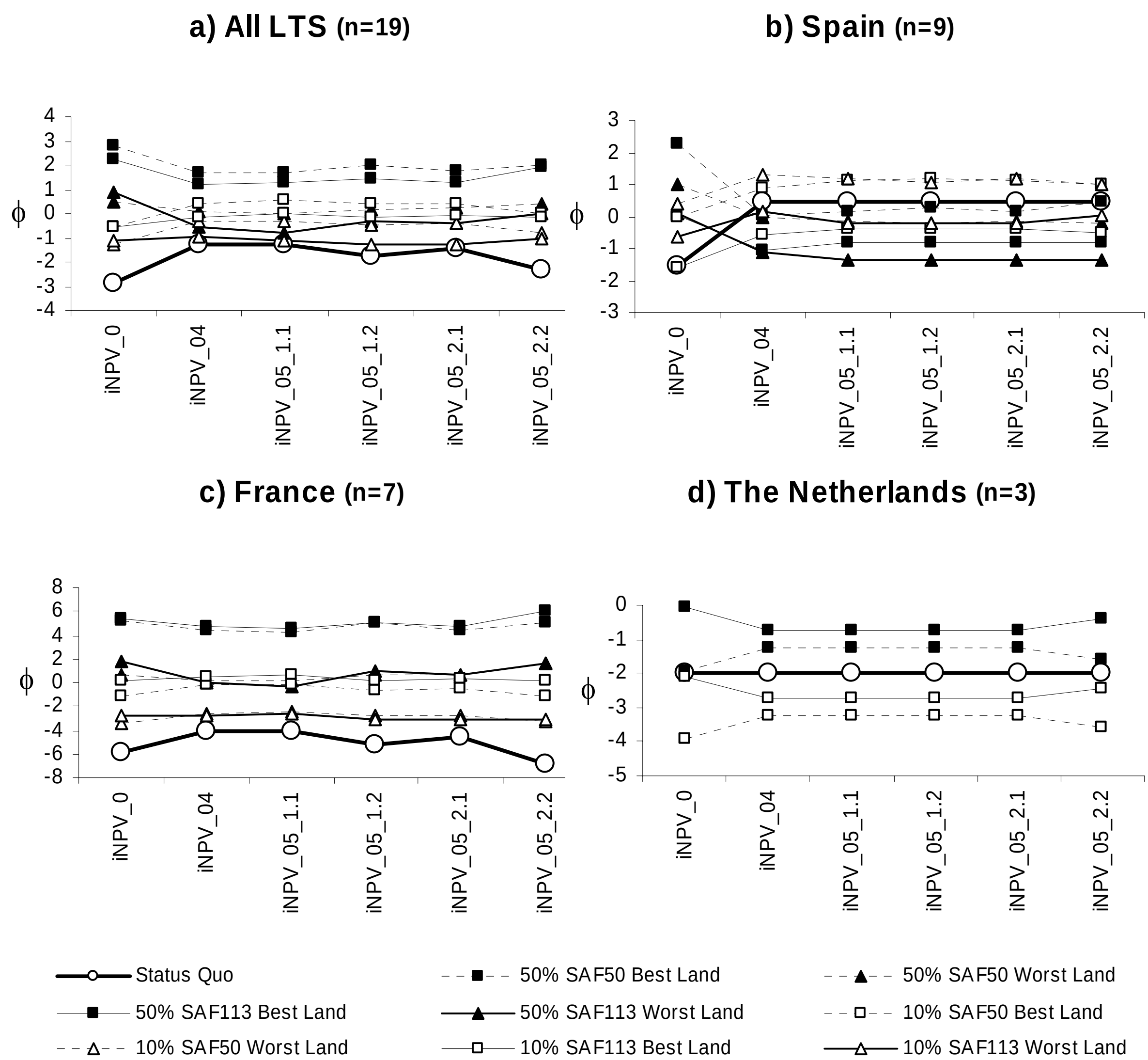


Figure 24: Preference rank of alternatives under six economic payment schemes a) for all landscape test sites (LTS) and b) for Spain, c) for France and d) for The Netherlands separately. The weight attribution represents an equilibrium evaluation between environmental and economic criteria (see section 5.3.4, page 93). Note: in The Netherlands there were only homogeneous landscapes, therefore only one quality of land was assessed.

In Spain, government policies on trees and crops favored conventional arable agriculture relative to agroforestry (*see section 4.4.4.2, page 74*) as shown by higher ranking of the status quo under government support (Figure 24b). Even so, the SAF alternatives with low tree density on 10% of the worst land were preferable to the status quo (Figure 24b – open symbols and dotted lines). In the absence of government support (Figure 24b – iNPV_0), SAF alternatives performed considerably better than the conventional arable alternative, especially in cases where 50% of the farms were converted to SAF.

In France, the final evaluation was less affected by differences in the payment schemes and depended more on the financial productivity of the systems. In fact, for all payment schemes, SAF alternatives performed better than the status quo. The best results were related to SAF implementation on 50% of the high quality land on the farm (Figure 24c – filled square symbols). The improved performance on high quality land is partly associated with the decision that high value walnut trees would tend to be allocated to such areas, whereas lower value wild cherry (*Prunus avium* L.) and poplar trees were generally allocated to lower quality land. Even so, on low quality land, wild cherry and poplar agroforestry systems often outranked the status quo arable cropping (Figure 24c).

In the Netherlands, land quality over the farm was homogenous. The SAF alternatives on 50% of the farm were preferable to the other alternatives (Figure 24d - filled symbols), and the relative ranking of arable cropping was not influenced by the subsidies as observed in Spain or France. However, the subsidies negatively affected the performance of SAF at higher tree densities (113 trees ha^{-1}) (Figure 24d – continuous lines), but this negative effect is recovered with tree payments in scenario iNPV_05_2.2. In this country, the performance value of the alternatives had a negative average (Figure 24d - yy axis) which means that none of the alternatives had dominance over the others and these had dominance over each alternative tested (Equations 19 and 20). In other words, the best performed alternatives were those which were less dominated by all the others.

5.5 Conclusions and further research

In MCDA the ecological impacts of SAF are usually assessed using qualitative approaches (e.g. Sipos, 2005). The evaluation presented here, used quantitative environmental and economic outputs as criteria in an integrated assessment of SAF in comparison with conventional agriculture.

The land-use alternatives were designed to reflect the questions that typically concern farmers, for example, regarding the proportion of the farm which should be allocated to the new land-use, the land quality which should be used, and the tree density of planting. The evaluation showed that in order to achieve environmental benefits, larger portions of the better quality land should be converted; tree density was a less important consideration (Figure 23). Economic performance was improved if high value trees were established on the highest quality farmland, rather than lower value trees on low quality land. Thus, the main conclusion to be drawn from these findings is that new SAF systems in Europe will generally have the greatest benefit if they are established over substantial areas of the farm and on the best quality land.

The intention of the alternative payment schemes was to derive recommendations on the type and level of public subsidies in order to render SAF (economically) competitive with conventional arable cropping and therefore to "harvest" its environmental benefits. However, the CAP reforms implemented in 2005 were predicted to have only a minimal effect on the profitability of SAF relative to the pre-reform situation (iNPV_04). For example, whether the SFP was available for the cropped area (iNPV_05_1.1) or the whole area (iNPV_05_1.2) made a difference only in France, where the competitiveness of SAF relative to the status quo was increased. Additional payments for tree establishment (schemes iNPV_05_2.1 and iNPV_05_2.2) made minimal difference in Spain and The Netherlands, while in France this effect was more important to the relative performance of the SAF and arable cropping alternatives.

Further research is needed to refine the approach presented here. For example, the financial analysis assumed that farmers would have access to capital that could be used for establishing SAF plantations, but lack of such access could limit SAF, especially in poorer areas of Europe. The analysis was also limited to a comparison of SAF and arable systems and a fuller analysis should include forestry systems. The evaluation could also be improved by weighting the selected criteria to reflect their relative importance in different regions or for different stakeholders (*section 5.3.4, page 93*). For example, in Spain, farmers may feel soil erosion control is more important than nitrogen leaching, which is rarely problematic in non-irrigated Mediterranean conditions, whereas in the Netherlands, nitrogen leaching is a major problem, but soil erosion is less of an issue (See Chapter 3).

In Europe, ecological integrity is increasingly seen as fundamental to economic and social well-being (Kay & Schneider, 1994). This paper has provided an initial approach for integrated environmental and economic analysis of SAF systems. Such results could be used to support policy development for SAF as a new land-use alternative for farmers (Lawson *et al.*, 2004, EC, 2005a, Lawson *et al.*, 2005).

Chapter 6

General conclusions

6.1 Environmental modelling framework

The modelling framework (Chapter 2) allowed the evaluation of the environmental assessment of SAF systems by the use of existing and state-of-art modelling tools.

The approach adapted the revised universal soil loss equation (Renard et al., 1997) to agroforestry scenarios, also taking into consideration related algorithms which tackle the problem of data availability at landscape scale. The nitrogen leaching assessment was undertaken trough simple algorithms (Feldwisch *et al.*, 1998) for which the nitrogen balance was calculated using European available databases (e.g. EMEP, 2003), literature data (e.g. Wild, 1993) and the inversion of the equations developed by van Keulen (1982) for estimating the quantity of nitrogen application. The approach was linked to the bio-physical dynamics of the agroforestry system modelled by Yield-SAFE (van der Werf *et al.*, 2006). Carbon sequestration was estimated using IPCC (1996) and Gifford (2000a) relationships which were linked to the Yield-SAFE predictions of tree biomass (Chapter 3 and 4) and a general evaluation of the effects of SAF implementation in the landscape biodiversity was estimated based on the algorithms developed in European landscapes (Bugter *et al.*, 2001).

In this first part of the work the methodology framework was developed. The most important activity in improving model predictions would be local validation of input and output data. Nevertheless, it is argued that, for the purpose of a broad assessment of the effects of SAF, the approach provides a balance between modelling complexity, the number of indicators and the geographic range under investigation.

6.2 Impact of SAF systems

The environmental modelling framework was applied to nineteen landscapes across Europe (Chapter 3). In the same landscapes, an economic model developed by a project partner was applied (Chapter 4). The selection of the landscape test sites covered a range from southern (Andaluzia - Spain) to northern Europe (Overijssel - The Netherlands).

The implementation of SAF was simulated through scenarios exploring the implementation of SAF in 10% or 50% of the farm/landscape, two densities of trees (50 and 113 trees ha^{-1}) and implementing the system in the best or worst quality of the land.

6.2.1 Environmental performance

The adoption of silvoarable agroforestry systems can potentially lead to reduced soil erosion and nitrogen leaching and increased carbon sequestration and landscape biodiversity. The extent of the modifications depends on severeness of problems at individual locations and on the management of the SAF system selected for each location. Predicted environmental benefits were highest when SAF was implemented on large areas (i.e. 50% of the farm) on high quality land, where current agricultural practices are most intensive and thus associated with higher levels of soil erosion and nitrogen leaching. Tree density (50 or 113 trees ha^{-1}) appeared less important, as in stands of lower density, biomass production per tree is higher, reducing the difference in values of the indicators on an area basis. The reduction in nitrogen leaching, however, was stronger at high tree densities. Because total tree biomass per area can have similar values, carbon is sequestered in the same relationship. The effect on landscape diversity was strongest in landscapes where agricultural monocropping is the dominating land-use type and where only few elements of ecological infrastructure exist.

6.2.2 Profitability

The "Land Equivalent Ratio" relates the area under sole cropping to the area under the agroforestry system, at the same level of management that gives an equal amount of yield. Under the economic conditions envisaged in the analysis, the financially most attractive systems tended to have a land equivalent ratio that was substantially above one. Conditions that most favoured a high land equivalent ratio appeared to be the use of relatively high tree-densities to make full use of available resources, the combination of deciduous trees with autumn-planted crops to make complementary use of light, and a high soil water availability to ensure that extra biomass production could be sustained. Conversely, it appeared that low ratios were associated with low tree density, evergreen trees, spring-planted crops, and low soil water availability. Under the hypothetical new CAP reform grants it was predicted that silvoarable systems with walnut and poplar in France could provide a profitable alternative to arable systems. In Spain, it appeared that holm oak and stone pine could be integrated into arable systems without significantly reducing arable production for many years. Since these trees are of ecological and landscape importance, rather than productive importance, additional support in the form of an agri-environment payment would be justified.

6.2.3 Integrated assessment

With the environmental and economic results assessed under the same biophysical and management regimes, the results could be truly compared (Chapter 5). Multi-criteria decision tools were used to evaluate the land use scenarios under a neutral preference between environmental and economic criteria.

For the majority of the scenarios in a general analysis, SAF was always preferred as it performed better considering all environmental and economic criteria. The best performing scenarios were those related to a higher area of implementation of SAF. However, under different country contexts, SAF might not be the best option because of payment incentives tends to favour conventional arable production.

When assessing the criteria independently, SAF systems performed better in the environmental outputs whilst the arable systems were generally more profitable except for some situations in France where SAF scenarios performed better than the arable in all environmental and economic criteria.

6.3 Revisiting the objectives

The first objective of this work was to provide a methodological framework to assess the environmental performance of SAF systems at farm/landscape scale. A set of modelling algorithms were adapted, and developed, providing the methodology to assess SAF systems within a European biophysical gradient.

The second objective was to develop a sampling strategy to allow an unbiased selection of test sites, derive SAF implementation scenarios and apply the environmental modelling framework. A range of test sites covering broad biophysical conditions were randomly selected with the help of European scale datasets. Scenarios were developed taking into consideration different implementation options. This allowed to evaluate the environmental performance of SAF in a European perspective.

The third objective was to combine the environmental and economic results into a single integrated assessment. This was achieved with the help of multi criteria decision tools which showed that considering all the criteria together, SAF systems can be a sustainable alternative to conventional agriculture.

6.4 Recommendations and outlook

In order to provide a broad assessment of the effect of SAF, this work had to balance scale and modelling complexity, the number of indicators used, the number of scenarios to evaluate and the geographic range of the study. The framework developed consists of a coherent approach to the integrated use of biophysical, economic, and environmental modelling tools at the appropriate spatial and thematic resolution. With this framework in place, the approach could be further improved by local validation of the input and output data. Also, beyond the application to agroforestry, the framework can be expanded to testing other alternative land-use systems such as new crops, agricultural energy fuel production, etc.

Silvoarable systems are complex, as there are many possible tree and crop arrangements, and implementation takes effect over a long period of time. Hence general scenarios were investigated for the provision of general guidelines. However, many other scenarios are possible, investigating for example different tree species, densities and arrangement, crop sequences, phased implementation of SAF on the farm and changes in management strategy over time in accordance with tree growth. All these options can be explored with the modelling approach and tools presented here.

Further research is also needed to evaluate stakeholders' preferences and assess which are the best alternatives/scenarios under their preferences. This could be assessed with other weight criteria distribution in the multi criteria decision analysis which could represent different stakeholders' interests.

Many factors such as risk, land ownership, family situation, and farmer's age can affect the choice of the most appropriate SAF system. The simulations indicate that SAF would often yield similar levels of profitability as conventional arable systems if there were no subsidies. However, under the pre- and post-2005 grant regimes, with the exceptions mentioned for France, the profitability of conventional arable systems was increased relative to SAF. As a consequence, under current payments, the uptake of SAF, if based on profitability alone, will be restricted to specific systems such as those examined in France, with high value walnut timber or fast growing poplar.

The approach and tools developed were applied in an integrated environmental and economic study designed to allow for an assessment of trade-offs between different indicators.

From the results, it appears that SAF justifies similar, if not greater, public support, than what is currently provided for conventional agricultural production.

With the recent CAP reform strengthening the emphasis on environmental performance and sustainable use of natural resources (EC, 2005b), SAF could play an important role since growing trees and crops in combination in SAF was found to be more productive than growing them separately in arable and forestry systems. However, many challenges in modelling and promoting SAF will need to be met if agricultural landscapes in Europe are again to benefit from the presence of widely-spaced trees.

The implementation of SAF at European scale had it first step in the council regulation no 1698/2005 on support for rural development by the European Fund for Rural Development (EC, 2005a) which contemplates financial support at European scale for the first time. These measures can potentially support the implementation of agroforestry experimental research sites in collaboration with farmers.

Silvoarable agroforestry is and will remain an interesting, challenging and fascinating research topic due to its complexity, which can cover many different aspects of multifunctional landscapes.

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Abbreviations

CAP – Common Agricultural Policy

LTS – Landscape Test Site

MCDA – Multi Criteria Decision Analysis

iNPV – Infinite Net Present Value

NPV – Net Present Value

RDR – Rural Development Regulation

SAF – Silvoarable Agroforestry

Summary

Silvoarable agroforestry (SAF) relates to the integration of trees and arable crops in the same field.

In Europe, agroforestry systems have been used mainly in traditional agriculture to provide a variety of agricultural and tree products. However, during the last three centuries, the agricultural landscape in Europe has seen a steady reduction of agroforestry. The reduction has been greatest since 1950, as the introduction of land consolidation programmes and agricultural mechanisation encouraged the removal of hedges and isolated trees from agricultural land. However, as the environmental costs of intensive agriculture have become apparent, there has been an increasing interest in the promotion of ecologically sound practices. Novel silvoarable techniques could potentially offer a range of environmental and economic benefits in comparison with conventional arable cropping.

The EU project "Silvoarable Agroforestry for Europe" investigated the European context of modern silvoarable agroforestry. It aimed to reduce uncertainties concerning the productivity and profitability of silvoarable systems, and to suggest European policy guidelines for agroforestry implementation.

Within this broad framework, an integrated assessment of the environmental and economic performance of SAF was undertaken with the objective of assisting decision-makers implement ecologically sound land management practices. This was done in four steps: (i) developing an environmental assessment platform, (ii) applying the platform to test sites across Europe, (iii) co-ordinating economic assessments to the same test sites, and (iv) joining both environmental and economic results into an integrated interpretation.

Chapter 2 describes the environmental assessment methodology to evaluate SAF systems based on the adaptation of existing models and the development of specific algorithms to predict the environmental effects of SAF at a farm- and landscape-scale. Modeling is needed when experiments are costly or too time consuming as in the case of agroforestry, where we assumed an average rotation to last 60 years. The framework comprised the assessment of soil erosion,

nitrogen leaching, carbon sequestration and landscape diversity and allowed the comparison of the environmental performance of SAF with arable monocropping using these four indicators. The method opted for a broad view of the four environmental indicators, is applicable over a large geographic range (from Mediterranean to Temperate Europe) and is based on spatial data that are generally available. The method was tested in three landscape test sites in Spain, in France and in The Netherlands. A total of five tree species were modelled: wild cherry (*Prunus avium* L.), black walnut (*Juglans* hybr.), poplar (*Populus* spp), holm oak (*Quercus ilex* L. subsp. *ilex*) and stone pine (*Pinus pinea* L.).

Chapter 3 systematically assesses the environmental effects of SAF for a stratified random sample of 19 landscape test sites (LTS) in Mediterranean and Atlantic regions of Europe. For each LTS, existing geographical and statistical data were compiled, harmonized and complemented by field surveys. LTS were subdivided into a maximum of four land units (LU) using cluster analysis. The LUs were considered to be homogenous with respect to soil properties and climatic conditions and were used to represent farm management units. The LUs were ranked according to their potential productivity from “best land” to “worst land”.The impact of SAF was explored by introducing SAF over 10% or 50% of the farm/landscape to simulate “pessimistic” and “optimistic” adoption by the farmer. Two tree densities (50 and 113 trees ha^{-1}) were compared and SAF could be implemented in the best or worst quality land of the LTS to simulate different management priorities.

Across the 19 landscape test sites, SAF had a positive impact on the four environmental indicators with the strongest effects when introduced on the best quality land. The computer simulations showed that SAF could significantly reduce erosion by up to 65% when combined with contouring practices. Nitrogen leaching could be reduced by up to 28% in areas where leaching currently is high (>100 kg N ha^{-1} a^{-1}), but this was dependent on tree density. With agroforestry, predicted mean carbon sequestration, over a 60-year period, ranged from 0.1 to 3.0 t C ha^{-1}a^{-1} (5 to 179 t C ha^{-1}) depending on tree species and location. Landscape biodiversity was increased by introducing SAF by an average factor of 2.6.

From the beginning of the research, Chapters 3 and 4 were co-ordinated and carried out in the same test sites. While Chapter 3 assessed the environmental effects of silvoarable agroforestry, Chapter 4 evaluated the profitability of the same scenarios as Chapter 3.

Time-series of annual production data and economic data for crops and trees (grants, revenue and costs) for each LTS were combined in the farm-scale bio-economic spreadsheet model FarmSAFE. The economic performance of the arable and silvoarable systems was compared using the infinite net present value (*iNPV*) for a time-frame of 60 years (discount rate = 4%).

The Common Agricultural Policy (CAP) payments were modelled for arable and silvoarable systems assuming: 1) No CAP payments; 2) Pre-2005 CAP payments; and 3) Post-2005 CAP payments, assuming in the case of silvoarable systems that a Single Farm Payments would be made to the whole cropped area (whilst cropping occurred) and that 50% of tree costs would be covered for the initial 4 years of the tree rotation.

The analysis in France suggests that walnut and poplar silvoarable systems could provide a profitable alternative to arable and forestry systems, while in Spain a modest restructuring of the amount and delivery of agricultural payments would increase the attractiveness of silvoarable systems of holm oak and stone pine. In the Netherlands, low timber value and the opportunity cost of losing arable land for slurry manure application made both silvoarable and forestry systems uncompetitive with arable systems.

Chapter 5 is the integration of results obtained in Chapters 3 and 4 into a multi-criteria decision analysis (MCDA). The PROMETHEE outranking approach was used to evaluate the integrated performance of silvoarable agroforestry relative to a status quo, on hypothetical farms in the nineteen LTS in Spain, France, and the Netherlands. The criteria used in the evaluation were soil erosion, nitrogen leaching, carbon sequestration, landscape biodiversity, and infinite net present value, the latter assessed under six levels of government support. The MCDA was not configured to reflect the position of a specific stakeholder (e.g. an NGO might rate environmental criteria higher than profitability, whereas farmers might rate profitability higher than environmental criteria) but a neutral weight distribution was adopted.

In France, the analysis showed, assuming equal weighting between environmental and economic performance, that silvoarable agroforestry was preferable to conventional arable farming. The best results were observed when agroforestry was implemented on 50% of the highest quality land on the farm; the effect of tree density (50-113 trees ha^{-1}) was small. By contrast, in Spain and the Netherlands, the consistently greater profitability of conventional arable agriculture relative to the agroforestry alternatives made overall performance of agroforestry systems dependent on the proportion of the farm planted, and the tree density and land quality used.

The environmental and economic performance of SAF in Europe is highly variable since each country/region has its specific biophysical and economic conditions. However, in Europe, ecological integrity is increasingly seen as fundamental to economic and social well-being. This work provided an initial approach for an integrated environmental and economic analysis of SAF systems. The findings could be refined to support policy development for silvoarable agroforestry as a new land-use alternative for farmers. At the same time, the framework could be adapted to the investigation of environmental and economic consequences of other land-use alternatives (e.g. new crops, forestation) at the landscape / farm scale.

Samenvatting

Boslandbouw met bomen en akkerbouwgewassen, in het vervolg kortweg aangeduid met SAF - “Silvoarable AgroForestry”, heeft betrekking op de integratie van bomen en akkerbouwgewassen in hetzelfde perceel.

In Europa komen boslandbouwsystemen met name voor in de traditionele landbouw om zo een spreiding in landbouwproducten en houtopbrengsten te geven. Echter in de laatste drie eeuwen laat het agrarische landschap in Europa een gestage vermindering van boslandbouw zien. De reductie is het grootst vanaf de jaren ’50, vanwege de introductie van landschapsbeheerprogramma’s en landbouwmechanisatie, die beide bijdroegen aan de verwijdering van houtwallen en alleenstaande bomen uit het agrarische landschap. Pas toen de milieukosten van intensieve landbouw zichtbaar werden, is er een toenemende interesse in de toepassing van ecologisch verantwoorde landbouw ontstaan. Nieuwe methodes van geïntegreerde boom-gewasteelten kunnen in principe een scala aan milieu en economische voordelen opleveren in vergelijking met de conventionele akkerbouw.

Het EU project “Silvoarable AgroForestry for Europe” heeft het Europese kader voor moderne boslandbouw met bomen en akkerbouwgewassen (SAF) onderzocht. Het had tot doel de onzekerheid met betrekking tot productiviteit en winstgevendheid van boom-gewassystemen te reduceren en om Europese richtlijnen voor het gebruik van boslandbouw voor te stellen.

In dit brede kader is een geïntegreerde beoordeling van de milieu en economische effecten van SAF gestart met het doel om bestuurders/beleidsmakers te assisteren bij het invoeren van een ecologisch verantwoord landbeheer. Dit is in vier stappen uitgevoerd: (i) ontwikkeling van een milieubeoordelingssysteem, (ii) toepassing van dit systeem op verschillende proefgebieden binnen Europa, (iii) afstemmen van economische beoordelingen binnen hetzelfde proefgebied en (iv) samenbrengen van zowel milieu als economische resultaten in een geïntegreerde beoordeling.

Hoofdstuk 2 beschrijft de methodologie om een milieubeoordeling van SAF systemen te evalueren op basis van een aanpassing van bestaande modellen en de ontwikkeling van speciale rekenmethodes om te komen tot een voorspelling van de milieueffecten van SAF op boerderij- en landschapsniveau. Modellen zijn met name van belang als experimenten duur zijn of te tijdrovend, zoals in het geval van SAF, waar we uitgaan van een gemiddelde rotatie van 60 jaar. Het raamwerk voor de beoordeling omvat bodemerosie, uitspoeling van stikstof naar het grondwater, koolstofvastlegging en landschapsdiversiteit. Dit raamwerk maakt het mogelijk om een vergelijking te maken van de milieueffecten van SAF en van monoculturen, gebruikmakend van de vier indicatoren. De methode, gekozen vanwege de brede inzet van de vier indicatoren, is toepasbaar op een groot geografisch gebied (van Mediterrane tot gematigde streken in Europa) en is gebaseerd op ruimtelijke, algemeen beschikbare gegevens. De methode is getest op drie proefgebieden met verschillende landschapstypen in Spanje, Frankrijk en Nederland. In totaal zijn vijf boomsoorten gemodelleerd: wilde kers (*Prunus avium* L.), zwarte walnoot (*Juglans* hybr.), populier (*Populus* spp), steeneik (*Quercus ilex* L. subsp. *ilex*) en pijnboom (*Pinus pinea* L.).

Hoofdstuk 3 behandelt systematisch de milieueffecten van SAF voor een willekeurige steekproef van 19 proefgebieden (LTS - landscape test sites) in Mediterrane and Atlantische gebieden in Europa. Voor ieder van deze proefgebieden zijn de bestaande geografische en statistische gegevens verwerkt, op elkaar afgestemd en aangevuld met veldgegevens. De proefgebieden zijn onderverdeeld in maximaal vier landschapstypen (LU - land units) gebruikmakend van cluster analyse. Er is aangenomen dat de landschapstypen homogeen zijn wat betreft de bodemeigenschappen en klimaat en binnen deze landschapstypen worden tevens verschillende agrarische bedrijfstypen onderscheiden. De landschapstypen zijn gerangschikt naar hun potentiële productie, van “beste” tot “slechtste” land. Het effect van SAF is onderzocht door SAF te introduceren op 10% of 50% van het agrarisch bedrijf/landschap om zodoende een “pessimistische” en een “optimistische” acceptatie van de boer/beheerder na te bootsen. Twee plantdichtheden (50 en 113 bomen/hectare) zijn vergeleken en SAF kon zowel op het beste als het slechtste land binnen de proefgebieden worden geïntroduceerd om zo verschillende prioriteiten in bedrijfsvoering te simuleren.

SAF had voor alle 19 proefgebieden een positief effect op ieder van de vier milieu-indicatoren, waarbij het grootste effect te zien was op het beste land. Computer simulaties lieten zien dat SAF erosie significant (tot 65%) kon reduceren wanneer het gecombineerd werd met een bedrijfsvoering die rekening hield met de landschapscontouren. Stikstofuitspoeling kon worden gereduceerd tot 28% in gebieden waar de actuele uitspoeling hoog (>100 kg N/hectare per jaar) is, maar dit hing af van de plantdichtheid van de bomen. Onder boslandbouw varieerde de voorspelde koolstofvastlegging over een periode van 60 jaar van 0.1 to 3.0 ton C/hectare per jaar (5 - 179 ton C/hectare), afhankelijk van de boomsoort en locatie. Na invoering van SAF nam de landschapsdiversiteit toe met een gemiddelde factor van 2.6.

Vanaf het begin van het onderzoek zijn de hoofdstukken 3 en 4 op elkaar afgestemd en is het onderzoek uitgevoerd op dezelfde proefgebieden. Terwijl in hoofdstuk 3 de nadruk lag op het beoordelen van de milieueffecten van SAF, is in hoofdstuk 4 de winstgevendheid van deze scenario's geëvalueerd.

Voor ieder proefgebied zijn tijdreeksen van jaarlijkse productie en economische gegevens voor gewassen en bomen (subsidies, opbrengsten en kosten) gecombineerd in een op boerderijniveau spelend bio-economisch spreadsheet model FarmSAFE. Daarbij is het economische effect van pure akkerbouw- en gecombineerde boom-gewassystemen geëvalueerd gebruikmakend van de zogenaamde Netto-Contante Waarde (NCW) over een oneindige horizon (Eng.: "infinite net present value" - *iNPV*) voor een periode van 60 jaar (discount rate = 4%).

De bijdragen vanuit de Common Agricultural Policy (CAP) zijn gemodelleerd voor zowel akkerbouw- als boom-gewassystemen onder de volgende aannames: 1) Geen CAP bijdrage; 2) Pre-2005 CAP bijdrage; en 3) Post-2005 CAP bijdrage. Bij deze laatste is aangenomen dat in geval van boom-gewassystemen "Single Farm Payments" zouden gelden over het gehele bebouwde oppervlak (dus met gewas) en dat 50% van de kosten voor de bomen in de eerste vier jaar van de boomrotatie zouden gedekt worden.

De analyses in Frankrijk suggereren dat boom-gewassystemen met walnoot en populier een winstgevend alternatief kunnen bieden aan pure akkerbouw- of bosbouwsystemen, terwijl in Spanje onder een eenvoudige aanpassing van de landbouwsubsidies boom-gewassystemen met steeneik en pijnboom aantrekkelijker worden. In Nederland zijn, ten gevolge van een lage houtprijs en het verlies van bouwland om giermest uit te rijden, zowel boom-gewas als bosbouwsystemen niet rendabel in vergelijking met akkerbouwsystemen.

Hoofdstuk 5 beschrijft de integratie van de resultaten uit de hoofdstukken 3 en 4 in termen van een multi-criteria analyse (MCDA). De PROMETHEE methode is gebruikt om de geïntegreerde effecten van SAF ten opzichte van de huidige situatie, op hypothetische boerderijen in de 19 proefgebieden in Spanje, Frankrijk en Nederland, te evalueren. De gebruikte criteria zijn bodemerosie, stikstofuitspoeling, koolstofvastlegging, landschapsbiodiversiteit en de Netto-Contante Waarde over oneindige horizon, waarbij de laatste beoordeeld is op basis van zes vormen van overheidssteun. De MCDA is niet zo uitgevoerd dat een specifieke betrokkene wordt geëvalueerd (bijv. een niet-gouvernementele organisatie (NGO) zou milieueffecten zwaarder kunnen laten meewegen dan winstgevendheid, terwijl boeren winstgevendheid boven milieueffecten zouden kunnen stellen), maar een analyse met weging van diverse gebruikers is toegepast.

Uit de analyse kwam dat in Frankrijk, onder de aanname dat milieu- en economische effecten even zwaar worden gewogen, SAF de voorkeur genoot boven conventionele akkerbouw. De beste resultaten werden gevonden voor boslandbouw op 50% van het beste land van het bedrijf; het effect van verschillende plantdichtheden (50-113 bomen/hectare) was klein. In Spanje en Nederland, daarentegen, maakte de systematische grotere winstgevendheid van conventionele akkerbouw ten opzichte van de alternatieven van boslandbouw het overall effect van SAF afhankelijk van het percentage land gebruikt voor SAF, de plantdichtheid van de bomen en de kwaliteit van het land.

De milieu- en economische effecten van SAF in Europa is sterk plaatsafhankelijk, omdat ieder land/regio zijn eigen specifieke biofysische en economische omstandigheden kent. Echter, in Europa, wordt ecologische integriteit steeds meer gezien als voorwaarde voor economisch en sociaal welzijn. Dit werk geeft een aanzet tot een geïntegreerde milieu- en economische analyse van SAF systemen. De bevindingen uit dit proefschrift kunnen worden verbeterd om verdere ondersteuning te geven bij de beheersontwikkeling van SAF als een nieuw alternatief van landgebruik voor boeren. Tegelijkertijd kan het raamwerk worden aangepast om de milieu- en economische effecten van ander landgebruik (bijv. nieuwe gewassen, bebossing) te onderzoeken op landschap/boerderijniveau.

PE & RC PhD Education Statement Form

With the educational activities listed below the PhD candidate has complied with the educational requirements set by the C.T. de Wit Graduate School for Production ecology and Resource Conservation (PE&RC) which comprises of a minimum total of 22 credits (= 32 ECTS = 22 weeks of activities).

Review of literature (4 credits)

- Integrated Assessment of Silvoarable Agroforestry at Landscape Scale (2002)

Writing of Project Proposal (4 credits)

- Integrated Assessment of Silvoarable Agroforestry at Landscape Scale (2002)

Post-Graduate Courses (3 credits)

- Wind and Water erosion: Modelling and Measuring (2003)
- Introduction to Programming in ArcObjects with VBA (2004)

Deficiency, Refresh, Brush-up and General courses (6 credits)

- Applied Landscape Ecology (2003)
- La gestion du Savoir sur L'UE (2003)
- Writing Scientific Papers (2004)
- Vbasic for Applications – Intermediate (2005)
- Vbasic for Applications – Expert (2005)

PhD discussion groups (6 credits)

- Workshop on Upscaling Pilot Study, Spain (2002)
- Workshop on Agroforestry Biophysical Modelling, Spain (2003)
- Workshop on Agroforestry Biophysical Modelling, UK (2004)
- Biannual SAFE project Meetings (05)

Annual meetings, seminars and Introduction days (2.75 credits)

- Umwelwissenschaft Tag (2002)
- 1st Ann. Symp.f the PhD Program in Agroecology, Zürich (2004)
- 2nd Ann. Symp. of the PhD Program in Sustainable Agriculture, Zürich (2005)

- Numerous seminars on agroecology held at Agroscope Reckenholz-Tänikon, Zürich (2002-2006)

International symposia, workshops and conferences (10 credits)

- 6th IFSA Symposium on Farming and Rural Systems Research and Extension, Portugal (2004)
- VIII Congress of the European Society of Agronomy, Denmark (2004)
- 1st World Congress of Agroforestry, USA (2004)
- IALE - Landscape Ecology in the Mediterranean, Portugal (2005)
- Int. Conf. on Sustainable Land Use in Intensively used Agricultural Regions, Germany (2005)
- 7th IFSA Symposium on New Visions for Rural Areas, The Netherlands (2006)

Curriculum vitae

João Henrique Nunes Palma was born in Lisbon (Portugal) on the 2nd of May 1975. In Lisbon, he accomplished the basic, secondary and complementary education in 1993. Then he moved to Évora where he finished the biology diploma degree in 1999. During this period he was member of the students association board, organized several seminaries on science and travel experiences and organized the 1st National Meeting of Biology Students and the 1st Congress of Biology Students. During 1997 he got a PRODEP fellowship where he developed and applied land-use GIS modelling in a region in southern Portugal. In 1999 he finished his diploma thesis on the extraction and comparison of essential oils of aromatic plants.

During a period of two years (2000-2001) he experienced non-scientific professional activities in Crawley (UK), Yarmouth (UK) and La Plagne (FR). In the UK he embraced challenges as data entry in banking and electricity services, serving as bartender, website designer and cleaning services. In France he was a chalet manager in the French Alps for a winter season.

In 2002 he started his PhD in Zürich at the Swiss Federal Research Station for Agroecology and Agriculture (Agroscope FAL Reckenholz) and engaged the Wageningen PhD sandwich program in cooperation with the Laboratory of Geo-Information Science and Remote Sensing and the Forest Ecology and Forest Management Ecology group. During his PhD he followed the training and supervision plan (see previous page) of the C.T de Wit Graduate School for Production Ecology and Resource Conservation.

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"Silvoarable agroforestry is and will remain an interesting, challenging and fascinating research topic due to its complexity, which can cover many different aspects of multifunctional landscapes. "