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

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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.

Keywords: Erosion, nitrogen leaching, carbon sequestration, landscape diversity, modeling, alley cropping

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.

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 by Graves *et al.* (2007).

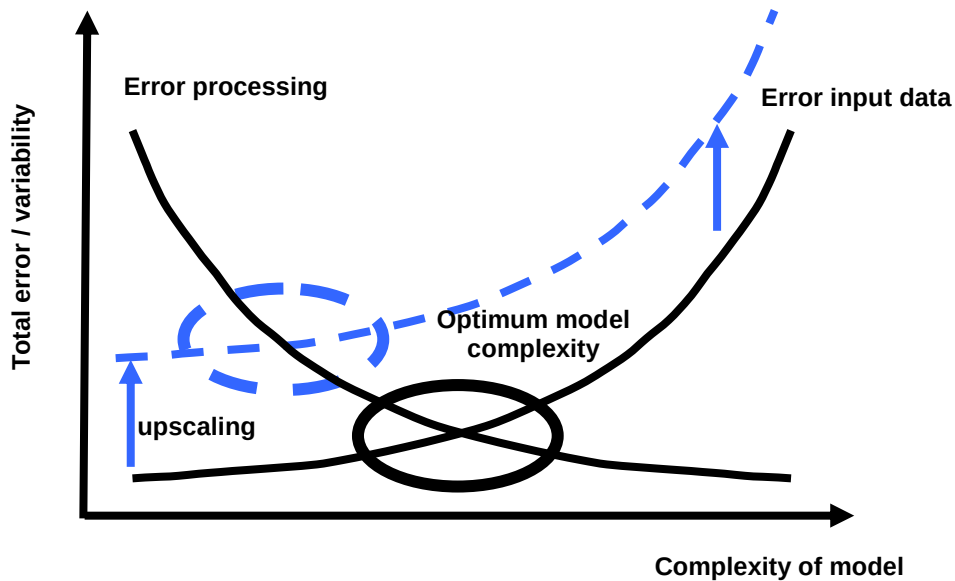


Figure 1: Relationship between model complexity and total error in the up-scaling process (Source: Wenkel & Schultz, 1999, modified).

Material and methods

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.*, 2007). 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).

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 (Graves *et al.* 2007).

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 (<i>Quercus ilex</i>)	Holm oak (<i>Quercus ilex</i>)	Wild cherry (<i>Prunus avium</i>)	Walnut (<i>Juglans hybr.</i>)	Poplar (<i>Populus spp.</i>)
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

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) (Equation 1):

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

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:

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

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.

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:

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

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).

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

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

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.*, 2007) which was parameterized and calibrated for the tree species and crop rotation at each LTS (Graves *et al.*, 2007).

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

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

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 $\text{kg N ha}^{-1}\text{a}^{-1}$.

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:

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

During the SAF rotation, tree growth and the later conversion to permanent grassland may disturb the N_{min} - I equilibrium through 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 (Graves *et al.*, 2007; van der Werf *et al.*, 2007). For a given crop and tree yield, the nitrogen uptake (U ; units: kg N ha^{-1}); was estimated as:

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

where: Y_c is the crop yield (unit: kg ha^{-1}); Y_{max} is the maximum crop yield (unit: kg ha^{-1}); B_t 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 B_t . The value of α is dependent on the biomass of the straw (S ; unit: kg ha^{-1}) and the harvested product (Y_c). A content of 1% and 0.4% N in the grain and straw was assumed respectively (van Keulen & Wolf, 1986) (Equation 8):

$$\alpha = \frac{1}{0.01 + 0.004 \frac{S}{Y_c}} \quad (\text{Equation 8})$$

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.

$$\lambda = 0.0066 + (0.0041 RSR) \quad (\text{Equation 9})$$

The fertilizer application was then estimated by:

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

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 ($30 \text{ kg N ha}^{-1} \text{ a}^{-1}$). 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 $1 \text{ kg N ha}^{-1} \text{ a}^{-1}$ (Wild, 1993).

Assessment of carbon sequestration

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

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

where, B_t is aboveground tree biomass (kg ha^{-1}), predicted by the Yield-SAFE model (Graves *et al.*, 2007).

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:

$$I_{\text{hab}} = \frac{A_{\text{hab}}}{A_{\text{total}}} \quad (\text{Equation 12})$$

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.

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) followed the same

status quo rotation, unless the crop component of the silvoarable system became unprofitable, in which case grass was assumed (Graves *et al.*, 2007). 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.*).

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.

Results and discussion

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.*, 2004), 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.

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⁻¹a⁻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.

By using the proportions of the different LU in each LTS (Table 1), 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).

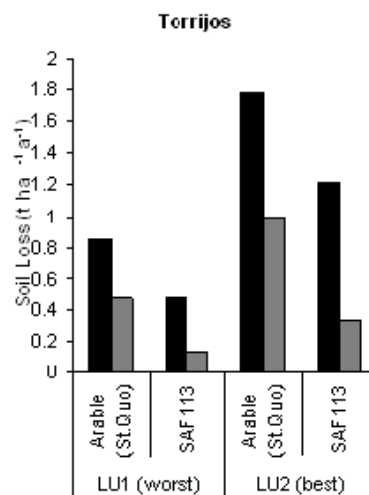


Figure 2: Estimated soil loss, at plot scale, for arable systems (St. Quo) and agroforestry (SAF₁₁₃) in the Torrijos landscape test site, central Spain. LU – Land Unit; See Table 1 for description of rotations.

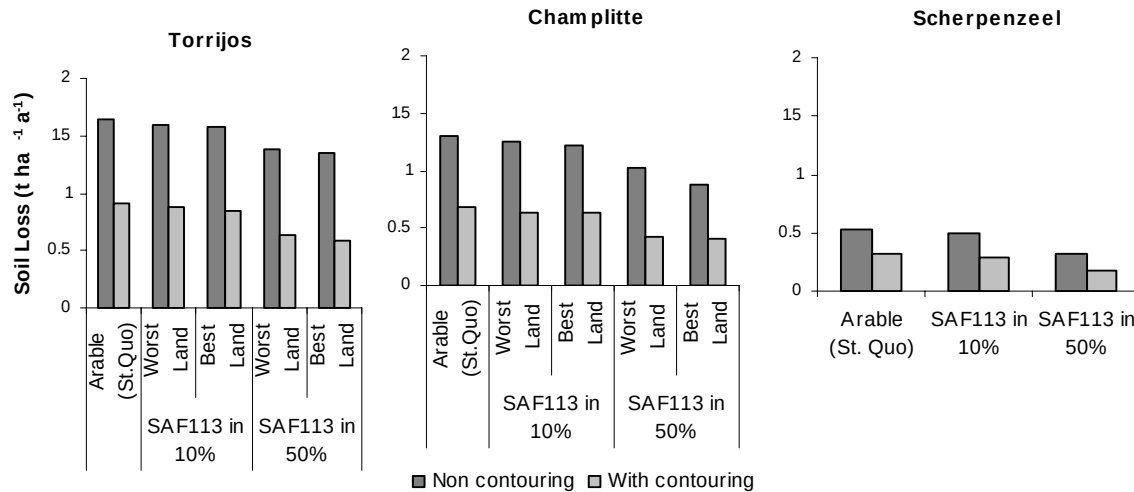


Figure 3: Estimated soil loss, at farm/landscape scale, for Torrijos (Spain), Champlitte (France) and Scherpenzeel (the Netherlands) for non-contouring and contouring practices. See Table 1 for crop rotations and tree species, section Scenarios for definition of scenarios.

Assessment of nitrate leaching

The assessment of nitrate 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.*, 2007) which was parameterized and calibrated for the selected tree and crop species in each LTS (Burgess *et al.*, 2005). For the low quality LU 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 m³ 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).

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. 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).

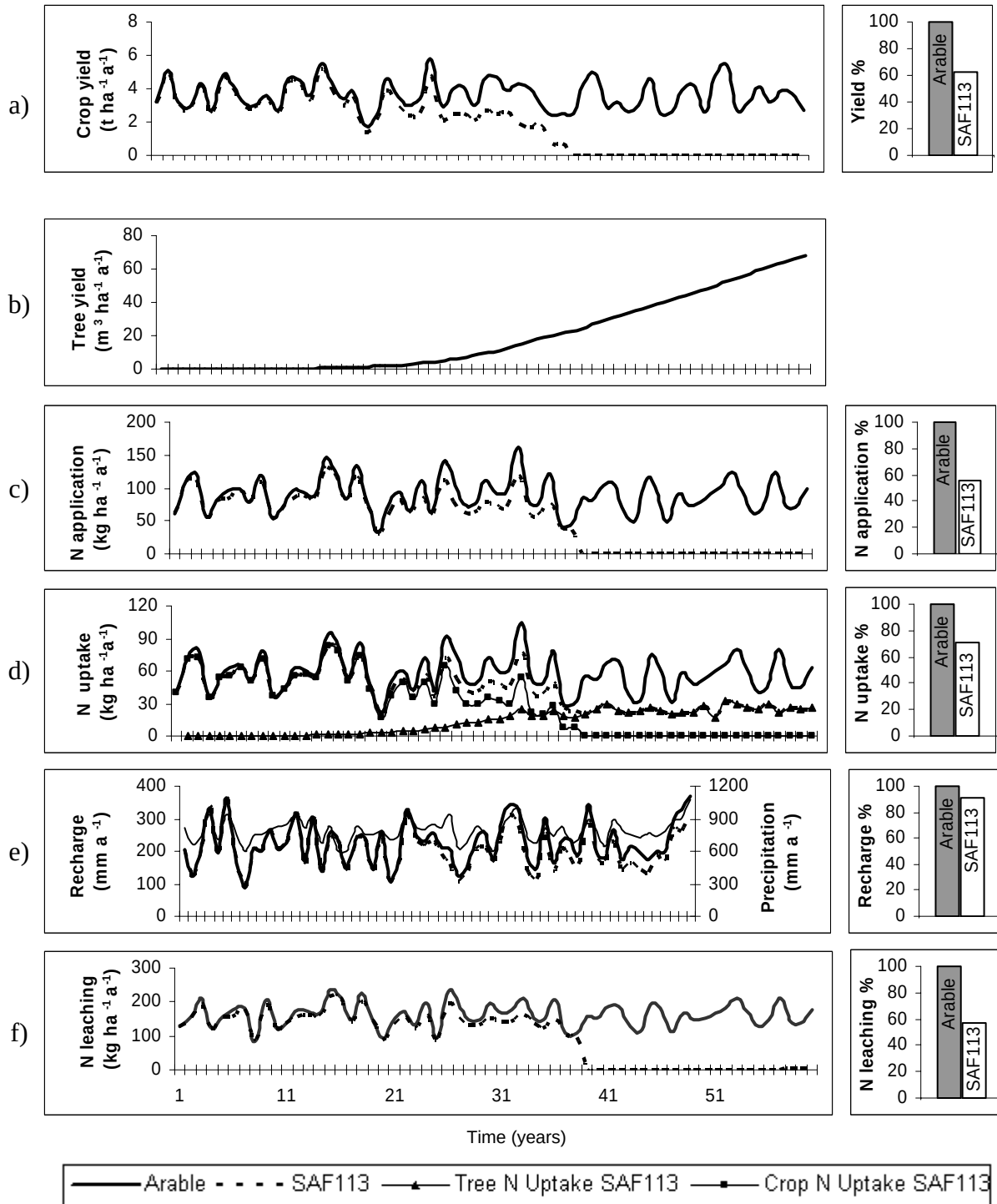


Figure 4: Comparison, at plot scale (LU2), between arable systems and agroforestry (SAF₁₁₃) in the Champlitte LTS, east France. Tree: wild cherry; crop rotation: Wheat-Wheat-Oilseed rape. Soil texture: medium; Soil depth: 35 cm. a) Crop yield; b) Tree yield; c) N application; d) N uptake; e) Precipitation and recharge; f) N leaching. Bar graphs: Relative cumulative results for 60 years.

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.

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).

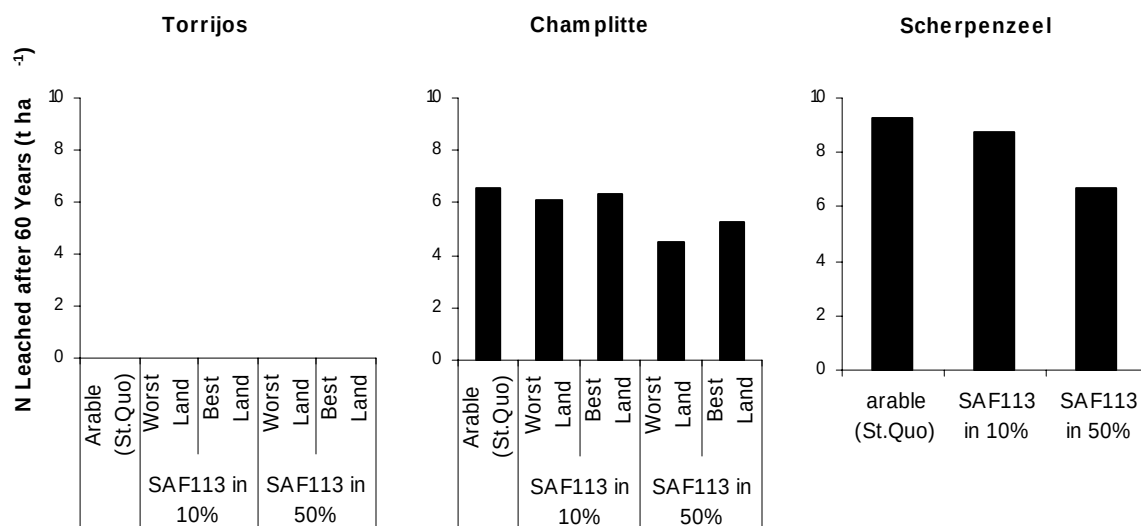


Figure 5: Estimated N leaching at the farm/landscape scale, cumulated over 60 years, for Torrijos (Spain), Champlitte (France) and Scherpenzeel (the Netherlands). Note the neglectable leaching in the mediterranean LTS due to lack of drainage. See Table 1 for crop rotations and tree species, section Scenarios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

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 (Graves *et al.*, 2007) and Equation 11. 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).

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.

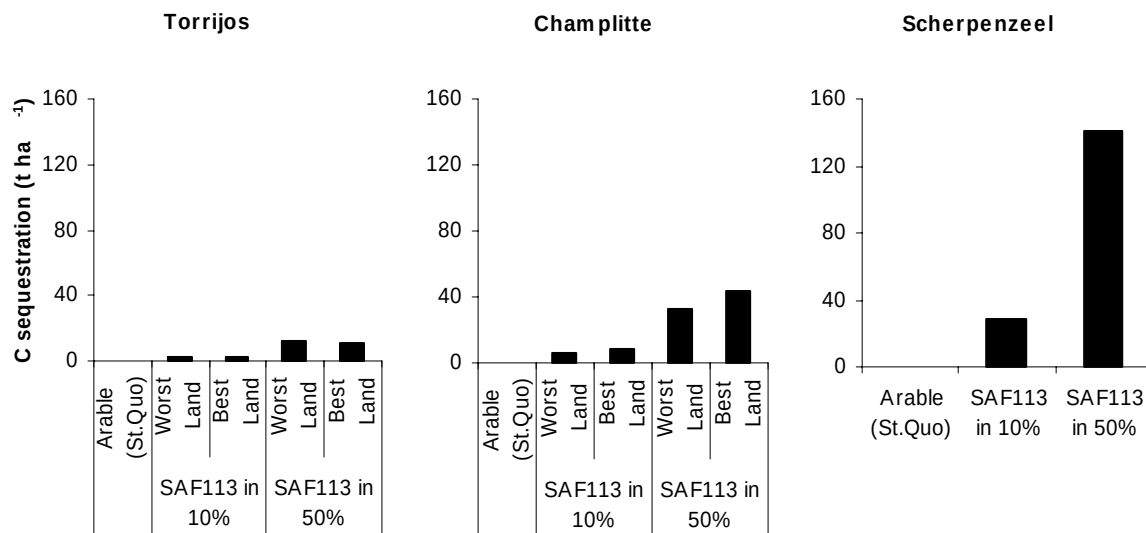


Figure 6: Estimated carbon sequestration at the farm/landscape scale, cumulated over 60 years, for Torrijos (Spain), Champlitte (France) and Scherpenzeel (the Netherlands). See Table 1 for crop rotations and tree species, section Scenarios for definition of scenarios.

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 below 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). 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%).

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).

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).

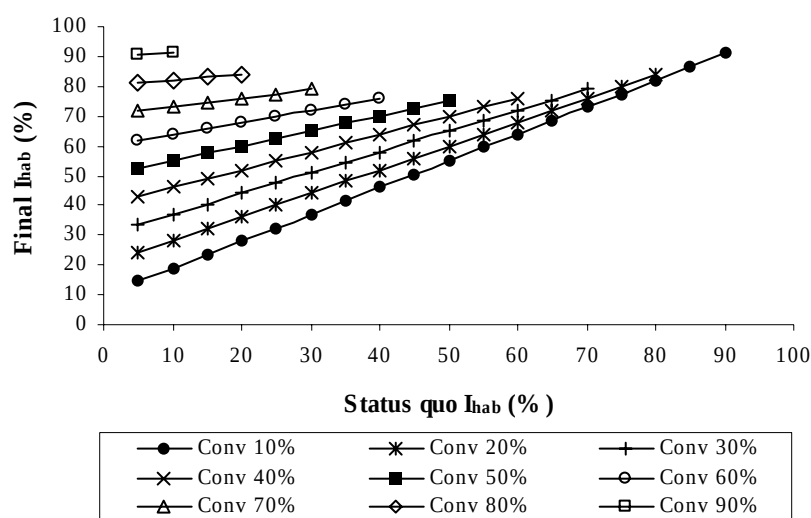


Figure 7: Relation between Status quo and Final natural and semi-natural habitat index (I_{hab}) by converting different proportions of arable land into agroforestry in the farm/landscape. See section Scenarios for definition of scenarios.

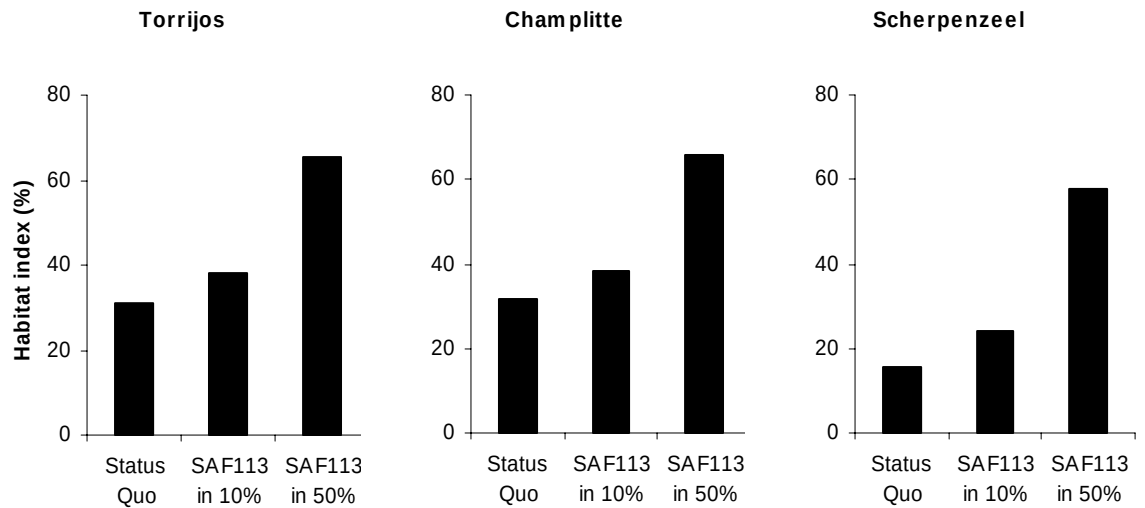


Figure 8: Estimated habitat index, at the farm/landscape scale, for Torrijos (Spain), Champlitte (France) and Scherpenzeel (the Netherlands). See section Scenarios for definition of scenarios.

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 (Graves *et al.*, 2007) to provide an integrated environmental and economic analysis of SAF.

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