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Modeling environmental benefits of silvoarable agroforestry in Europe

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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 the Mediterranean and Atlantic regions of Europe. 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* spp., 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 $ha^{-1}a^{-1}$ (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. The implications of this potential for environmental benefits at European scale are discussed.

Keywords: Alley cropping, carbon sequestration, erosion, landscape diversity, land use, nitrogen leaching, agri-environmental policy

1 Introduction

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

Increased agricultural output per unit of area and per unit of labor has been achieved using improved genetic material, increased inputs, and modern management techniques, 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. To some extent, however, the gains in efficiency of production were achieved at the expense of the environment. In many places, semi-natural and natural habitats were removed, resulting in reduced farmland biodiversity. Soil erosion and compaction and the pollution of ground- and surface water with nitrates and pesticides are other undesirable consequences of modern, intensified agricultural production (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 between the two components. 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 strongly declined because of agricultural intensification (Dupraz & Newman, 1997, Herzog, 1998).

In Europe, environmental benefits are expected from new land-use systems (Baldock et al., 1993). Since the 1990's, research projects have demonstrated that novel temperate agroforestry systems can operate 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). 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 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).

A modeling approach was developed by Palma et al. (2006) to assess the environmental performance of SAF systems. It comprised 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 Yield-SAFE (from "YIeld Estimator for Long term Design of Silvoarable AgroForestry in Europe"). Yield-SAFE was developed with as few equations and parameters as possible to allow model parameterization under constrained availability of data from long term experiments (van der Werf et al., 2006).

The objective of this paper is 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) at the scale of farms / small landscapes using models and algorithms of appropriate spatial and thematic resolution and complexity (Palma et al., 2006).

2 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. Yield-SAFE (van der Werf et al., 2006) was used to generate crop yields for typical crop rotations (combinations with up to five crops; wheat *Triticum* spp., sunflower *Helianthus annuus* L., oilseed rape *Brassica napus* L., grain maize and silage maize *Zea mays* L.) at each LTS over a 60-year time horizon. The same crop rotations were then incorporated in SAF systems that included holm oak (*Quercus ilex* subsp. *ilex* L.) and stone pine (*Pinus pinea* L.) in Spain, 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 Yield-SAFE model and environmental assessment algorithms.

2.1 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 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 (**Error! Reference source not found.**).

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 (Bugter et al., 2001). Digital elevation models were acquired from DLG© and digital soil maps from GeoDesk©. For 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 Yield-SAFE. 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, unshaded 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).

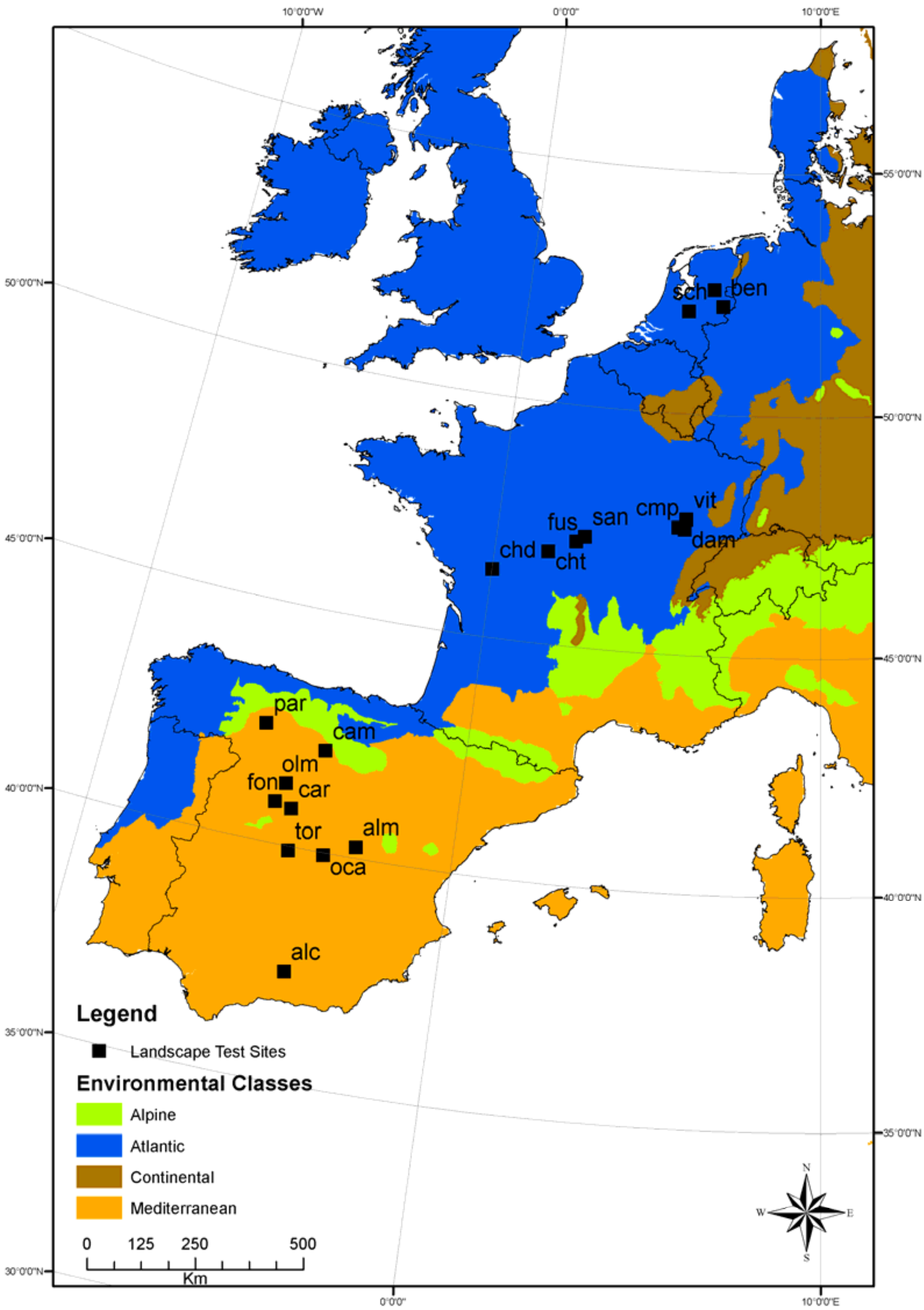

Fig 1. Landscape test sites selected covering wide biophysical characteristics based on the European environmental classification (Metzger et al., 2005). See Table 1 for the site codes.

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 clusters or land units (LU). Each LU was then characterized by its mean radiation and its major soil texture and soil depth (Fig 1). The cluster analysis resulted in 42 LU for the 19 LTS, each potentially producing different tree and crop yields (Table 1). All analyses, except that for landscape biodiversity, were restricted to agricultural land within 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 1).

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 (Fig 1).

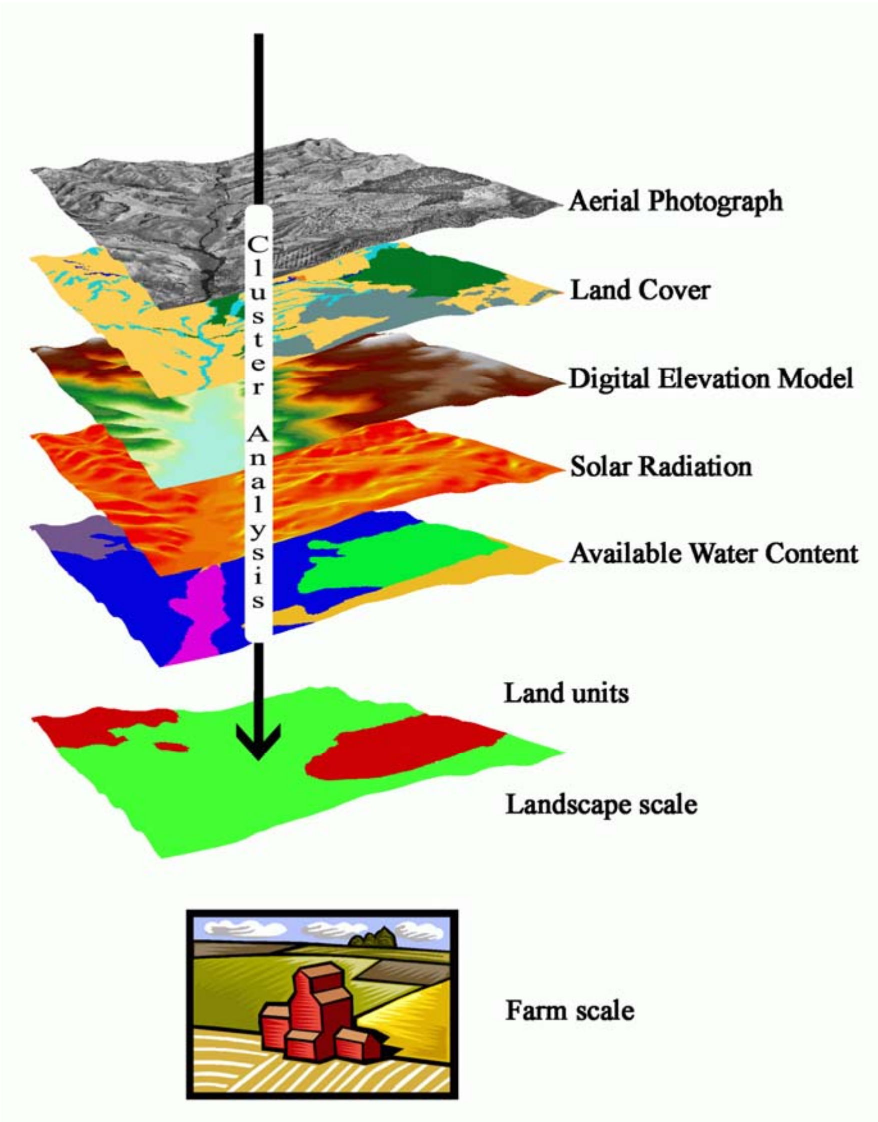

Fig 1: GIS landscape data processing for each landscape test site (LTS) to create homogeneous land units (LU), corresponding to different qualities of agricultural land of a farm (Torrijos LTS example).

Table 1: 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.

2.2 *Biophysical modeling*

The radiation, temperature, rainfall, soil depth and texture data for each LU 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 (e.g yield tables) 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 that light and temperature but not water, limited growth (Burgess *et al.*, 2004a). Then at each LU 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 the 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 LTS 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 LTS in year 60 were assumed to be 0.22 m^3 and 0.26 m^3 $tree^{-1}$ 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 (non-irrigated) 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.

2.3 *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 by Graves et al. (2006).

For each scenario, soil erosion, nitrogen leaching, carbon sequestration and landscape biodiversity were examined using the method described by Palma et al. (2006). 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, taking into account nitrogen contents of crop-tree biomass, of the soil and the nitrogen recovery capacity by crops (van Keulen, 1982). Crop and tree yields were computed by Graves et al. (2006) and van der Werf et al. (2006) for each LU using Yield-SAFE. The model also provided an estimation for 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 Yield-SAFE model. A broad evaluation of the effects of SAF implementation on landscape biodiversity was conducted, based on the share of habitats available to wildlife in an agricultural landscape, classifying each LTS into "habitat" (e.g. hedgerows, permanent grassland, traditional orchards) and "non-habitat" (the arable matrix).

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 LTS representing a hypothetical farm. These were then aggregated to provide an overall assessment of environmental effect for each scenario at each LTS / farm.

Modeled 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 Results and discussion

The resulting environmental assessments for each scenario in each LTS are summarized in Table 2. 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 (Fig 2).

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 order of magnitude of the computed values.

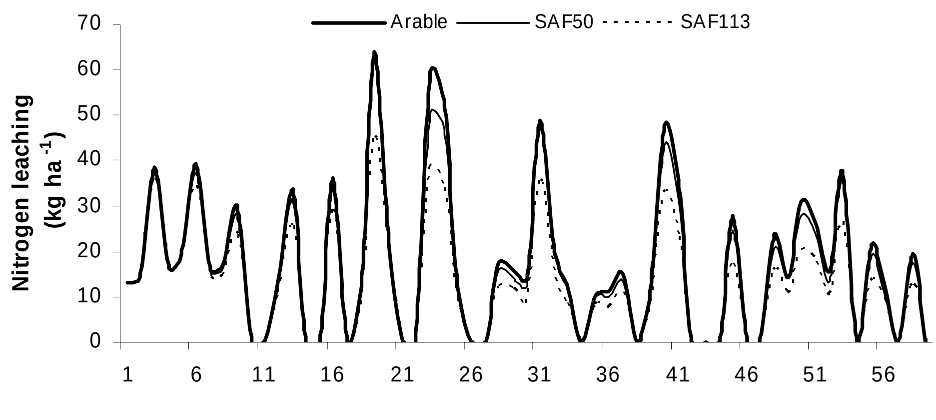

Fig 2: Predicted annual nitrate leaching at land unit scale in the Chateauroux landscape test site over a 60-year period with walnut and a wheat-wheat-oilseed rotation. Arable reference scenario (Arable, average annual leaching 17 kg N ha^{-1} a^{-1}), silvoarable scenario with 50 trees per hectare (SAF50, 16 kg N ha^{-1} a^{-1}), silvoarable scenario with 113 trees per hectare (SAF113, 12 kg N ha^{-1} a^{-1}).

Table 2: Projected effect of the introduction of silvoarable agroforestry (SAF) on erosion, nitrate leaching, carbon sequestration and landscape biodiversity at farm/landscape scale for different tree densities (50 and 113 trees ha^{-1}) on 10 and 50 % of the farmland on two land qualities.

Indicator	Scenarios: Erosion control practices	SAF density trees ha^{-1}	SAF Area (%)	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
		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
				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
			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
				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
		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
				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
			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
				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
	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
		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
				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
			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
				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
		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
				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
			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
				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
		SAF 50	10	Worst	0	0	0	0	0	0	1	7	0	37	71	49	61	132	107	136	151	103	131
				Best	0	0	0	0	0	0	1	7	0	37	70	51	59	137	106	133	151	103	131
			50	Worst	0	0	0	0	0	0	1	6	0	37	76	49	68	119	97	140	135	81	113
				Best	0	0	0	0	0	0	1	6	0	36	74	60	61	136	91	130	135	81	113
		SAF 113	10	Worst	0	0	0	0	0	0	1	6	0	36	70	47	60	131	102	132	146	99	125
				Best	0	0	0	0	0	0	1	7	0	36	70	49	59	126	105	127	146	99	125
			50	Worst	0	0	0	0	0	0	1	5	0	33	70	45	60	113	75	118	112	74	100
				Best	0	0	0	0	0	0	1	6	0	33	69	50	59	108	88	99	112	74	100

Table 2 (continued)

Indicator	Scenarios				LANDSCAPE TEST SITES																		
	Erosion control practices	SAF density trees ha^{-1}	Area (%)	Land quality	SPAIN									FRANCE							NETHERLANDS		
					ALC	TOR	OCA	ALM	CAR	FON	OLM	CAM	PAR	CHD	CHT	FUS	SAN	DAM	CMP	VIT	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 1 for landscape test sites codes.

3.1 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 2). 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 indicative of 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 2). To test significance, we grouped the LU and LTS into categories of low (< 0.5 t ha^{-1} a^{-1}), medium (0.5 - 3 t ha^{-1} a^{-1}) and high (> 3 t ha^{-1} a^{-1}) erosion sites (Table 3). 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. Only when both measures were combined, statistical analysis suggests significant reductions in erosion at medium (0.5 - 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 3a). Approximately the same order of 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 quality land.

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 3b). 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 the results from being statistically significant. Similar relative reductions in erosion rates have been found after the 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, implementing 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, the orientation of tree lines along the contour level would avoid this negative impact (Seobi et al., 2005). An aspect, which we did not investigate, is the risk of land slide processes, particularly important in slopes with soils consisting of layered clays. Although not modeled, the presence of trees in such areas could lower the risk of land sliding (Sidle et al., 2006).

Table 3: 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 b)on 10 and 50% of the worst or best quality land at the 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

3.2 Nitrate leaching

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 precipittion (Lehmann & Schroth, 2003). Such patterns of rainfall, typical of Mediterranean areas, were found at the Spanish LTS (Table 1), where for the arable status quo, nitrogen leaching was at most minimal (Table 2 – 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 2). 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 2). 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 2 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 10 kg ha^{-1} a^{-1}. The result for the LU (Table 4a) 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 4: 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	7	6	5	3	2	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 4b), 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 4b).

High nitrogen leaching was generally associated with high crop yields and high fertilizer application rates. At such sites, the Yield-SAFE model predicted greater tree-crop competition resulting in stronger crop yield reductions. Reducing the annual fertilizer applications in order to match the decreasing yield potential and ceasing cropping at an earlier point of the 60 year rotation of the system because intercropping was no longer profitable were the main causes of predicted reductions of nitrogen leaching. These effects were less pronounced at LTS with medium levels of nitrogen leaching, where nitrogen fertilizer application was less intensive and consequently, implementation of SAF did not significantly reduce nitrogen leaching (Table 4b).

The predicted reduction in nitrogen leaching under SAF appears conservative, compared to reported values of 40 to 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 for 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.

3.3 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 the soil due to leaf fall (Dixon, 1995, Montagnini & Nair, 2004) and in the vegetation strip along the tree line, these processes were not considered and therefore our values can be considered conservative. For the estimates, belowground tree biomass was estimated from the aboveground tree biomass calculated by Graves et al. (2006) and van der Werf et al. (2006), using allometric relationships (Palma et al., (2006). 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 2). 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 statistically significant differences between tree densities (Table 5a). Lower tree densities result in higher biomass per tree (Balandier & Dupraz, 1998, Graves et al., 2006, van der Werf et al., 2006), somewhat compensating for the low tree density. However, for slow growing trees, carbon sequestration is significantly higher for high tree densities (Table 5a) and when SAF is implemented in a large portion (50%) of the farm (Table 5b). With medium-fast growing tree species, these significant differences do not occur due to a higher variability of carbon sequestration of medium (hybrid walnut and wild cherry) and fast growing trees (poplar). No differences in sequestration were found between SAF systems established on high or low quality land, although sequestration was consistently somewhat higher on the best land. At farm-scale, significant differences in sequestration were only found between SAF on 10 or 50% of the farm (Table 5b).

Table 5: 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	15	15	8	7	7	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.

3.4 Landscape biodiversity

The habitat index (I_{hab}; 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 hypothetical farms were representative of land use in the LTS.

The introduction of SAF increased I_{hab} for all LTS, with the largest increase in areas of low shares of existing natural or semi-natural habitat. The current I_{hab} of the LTS varied from low values (I_{hab} < 10%) in homogeneous agricultural landscapes (e.g. Table 2 – OCA, CHT, BAL) to high values (I_{hab} > 60%) in more heterogeneous areas (e.g. Table 2 – 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 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. Our investigation addresses the potential extent of those environmental benefits of SAF systems, also including 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 farmland. 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). The use of medium-fast growing tree species in SAF systems, when implemented on 50% of the agricultural land, could contribute 46 - 96 t C ha^{-1} (0.77 - 1.6 t C $ha^{-1}a^{-1}$) to sequestration over a 60-year period. However, values up to 179 t C ha^{-1} (3 t C $ha^{-1}a^{-1}$) 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, this 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.

4 Conclusion

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 characteristics of individual sites and the management of the SAF system proposed for each location. Predicted environmental benefits were highest when SAF was implemented on large areas (i.e. 50% of the farm / landscape) 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.

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 by Graves et al. (2006) for the same test sites. Further analysis will address the integrated economic and environmental evaluation of the benefits and drawbacks of SAF systems.

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References

Alegre, J., Krishnamurthy, L., and Callo-Concha, D., 2004. Carbon Sequestration by Amazonian Agroforestry Systems, 1st World Congress of Agroforestry - Book of Abstracts. Orlando, USA, University of Florida - Institute of Food and Agricultural Sciences, p. 162. Available at: http://conference.ifas.ufl.edu/WCA/

Auclair, D., and Dupraz, C., 1998. Untitled - Preface. Agroforestry Systems. 43, 1-4.

Balandier, P., and Dupraz, C., 1998. Growth of widely spaced trees. A case study from young agroforestry plantations in France. Agroforestry Systems. 43, 151-167.

Baldi, A., Batary, P., and Erdos, S., 2005. Effects of grazing intensity on bird assemblages and populations of Hungarian grasslands. Agriculture Ecosystems & Environment. 108, 251-263.

Baldock, D., Beaufoy, G., Bennet, G., and Clark, J., 1993. Nature conservation and new directions in the EC common agricultural policy. HPC, Arnhem

Ball, G. H., and Hall, D. J., 1965. A Novel Method of Data Analysis and Pattern Classification. Stanford Research Institute, Menlo Park, California

Bolaños, F., Garcia del Barrio, J., González Ávila, S., and Elena-Roselló, R., 2003. REDPARES - Red de Paisajes Rurales Españoles - una herramienta básica para el estudio de los paisajes forestales de España. Montes. 73, 50-58.

Bouma, J., Varallyay, G., and Batjes, N. H., 1998. Principal land use changes anticipated in Europe. Agriculture Ecosystems & Environment. 67, 103-119.

Bugter, R. J. F., Burel, F., Cerny, M., Edwards, P. J., Herzog, F., Maelfait, J. P., Klotz, S., Simova, P., Smulders, R., and Zobel, M., 2001. In: (Editors), 92., pp., 2001. Vulnerability of biodiversity in the agro-ecosystems as influenced by green veining and land-use intensity: the EU project GREENVEINS. *in* Mander, U., Printsmann, A., and Palang, H., (Eds.), Development of European landscapes. Proceedings of the International Association for Landscape Ecology. Tartu, Publicationes Instituti Geographici Universitatis Tartuensis 92, p. 632-637.

Burgess, P., Graves, A., Metselaar, K., Stappers, R., Keesman, K., Palma, J., Mayus, M., and van der Werf, W., 2004a. Description of Plot-SAFE Version 0.3. Unpublished. Cranfield University, Silsoe, UK, 52 pp.

Burgess, P. J., 1999. Effects of agroforestry on farm biodiversity in the UK. Scottish Forestry. 53, 24-27.

Burgess, P. J., Incoll, L. D., Corry, D. T., Beaton, A., and Hart, B. J., 2004b. Poplar (Populus spp) growth and crop yields in a silvoarable experiment at three lowland sites in England. Agroforestry Systems. 63, 157-169.

Burgess, P. J., Incoll, L. D., Hart, B. J., Beaton, A., Piper, R. W., Seymour, I., Reynolds, F. H., Wright, C., Pilbeam, D. J., and Graves, A. R., 2003. The Impact of Silvoarable agroforestry with poplar on farm profitability and biological diversity. Final Project report to DEFRA. Available at: http://sciencesearch.defra.gov.uk/Document.aspx?DocumentID=641

Centeri, C., 2003. In situ soil erodibility values versus calculations. In: Gabriels, D., and Cornelis, W. (Eds.), 25 Years of Assessment of Erosion - International Symposium, International Center for Eremology - University of Ghent, Ghent, pp. 135-140.

Conrad, O., 1998. Derivation of Hydrologically Significant Parameters from Digital Terrain Models. Thesis. Dept. for Physical Geography. University of Göttingen, Göttingen. Available at: http://www.geogr.uni-goettingen.de/pg/saga/digem/

Di, H., and Cameron, K., 2002. Nitrate leaching in temperate agroecosystems: sources, factors and mitigating strategies. Nutrient Cycling in Agroecosystems. 64, 237-256.

Dixon, R. K., 1995. Agroforestry Systems - Sources Or Sinks Of Greenhouse Gases? Agroforestry Systems. 31, 99-116.

Dixon, R. K., Winjum, J. K., Andrasko, K. J., Lee, J. J., and Schroeder, P. E., 1994. Integrated Land-Use Systems - Assessment Of Promising Agroforest And Alternative Land-Use Practices To Enhance Carbon Conservation And Sequestration. Climatic Change. 27, 71-92.

Dupraz, C., and Newman, S., 1997. Temperate Agroforestry: The European Way. In: Gordon, A., and Newman, S. (Eds.), Temperate Agroforestry Systems, CAB International, Cambridge, pp. 181-236.

EC, 2003. FADN - Farm Accountancy Data Network, European Commission. Available at: http://www.europa.eu.int/comm/agriculture/rica/dwh/index_en.cfm

ECCP, 2003. Working Group on Forest Sinks - Final Report - Conclusions and Recommendations regarding forest related sinks & climate change mitigation. EU Comission, Brussels. Available at: http://europa.eu.int/comm/environment/climat/pdf/forest_sinks_final_report.pdf

Eichhorn, M., Paris, P., Herzog, F., Incoll, L., Liagre, F., Mantzanas, K., Mayus, M., Moreno Marcos, G., and Pilbeam, D., 2005. Silvoarable agriculture in Europe - past, present and future. Agroforestry Systems. (in press).

Ersahin, S., 2001. Assessment of spatial variability in nitrate leaching to reduce nitrogen fertilizer impact on water quality. Agricultural Water Management. 48, 179-189.

Feldwisch, N., Frede, H., and Hecker, F., 1998. Verfahren zum Abschätzen der Erosions und Auswaschungsgefahr. In: Frede, H., and Dabbert, S. (Eds.), Handbuch zum Gewässerschutz in der Landwirtschaft, Ecomed, Landsberg, pp. 22-57.

Forman, R., and Godron, M., 1986. Landscape Ecology. John Wiley & Sons, New York - Chichester - Brisbane - Toronto - Singapore, 619 pp.

GDS, 2005. Database of historical climate data compiled by Global Data Systems, United States Department of Agriculture World Weather Board from World Meteorological Organisation climate reporting systems. Available at: http://hydrolab.arsusda.gov/nicks/nicks.htm
Gifford, R., 2000a. Carbon Content of Woody Roots: Revised Analysis and a Comparison with Woody Shoot Components. National Carbon Accounting System Technical Report No. 7 (Revision1). Australian Greenhouse Office, Canberra, 10 pp.
-, 2000b. Carbon Contents of Above-Ground Tissues of Forest and Woodland Trees. National Carbon Accounting System, Technical Report No. 22. Australian Greenhouse Office, Canberra, 24 pp.
Graves, A., Burgess, P., Palma, J., Herzog, F., Moreno, G., Bertomeu, M., Dupraz, C., Liagre, F., Keesman, K., and van der Werf, W., 2006. The development and application of bio-economic modelling for silvoarable systems in Europe. Ecological Engineering. Accepted.
Griffiths, J., Phillips, D. S., Compton, S. G., Wright, C., and Incoll, L. D., 1998. Responses of slug numbers and slug damage to crops in a silvoarable agroforestry landscape. Journal Of Applied Ecology. 35, 252-260.
Grübler, A., 1994. Technology. In: Meyer, W., and Turner, B. (Eds.), Changes in Land Use and Land Cover, Cambridge University Press, Cambridge, pp. 287-328.
Herzog, F., 1998. Streuobst: a traditional agroforestry system as a model for agroforestry development in temperate Europe. Agroforestry Systems. 42, 61-80.
Herzog, F., Reisner, Y., and Palma, J. H. N., 2004. Working visit report on upscaling for nine landscape test sites in Spain. Agroscope FAL Reckenholz. Unpublished, Zurich.
Hoffmann, M., and Johnsson, H., 2003. Test of a modelling system for estimating nitrogen leaching - A pilot study in a small agricultural catchment. Environmental Modeling & Assessment. 8, 15-23.
IPCC, 1996. Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories: Reference Manual. Available at: http://www.ipcc-nggip.iges.or.jp/public/gl/guidelin/ch5ref3.pdf
Klaa, K., Mill, P. J., and Incoll, L. D., 2005. Distribution of small mammals in a silvoarable agroforestry system in Northern England. Agroforestry Systems. 63, 101-110.
Kleijnen, J., 1987. Statistical tools for simulation practicioners. Marcel Dekker Inc, New York - Basel, 429 pp.
Kürsten, E., 2000. Fuelwood production in agroforestry systems for sustainable land use and CO2-mitigation. Ecological Engineering. 16, S69-S72.
Lane, L., and Nearing, M., 1995. USDA-Water Erosion Prediction Project (WEPP):Hillslope Profile and Watershed Model Documentation. NSERL Report #10. USDA-ARS, West Lafayette, 269 pp. Available at: http://topsoil.nserl.purdue.edu/nserlweb/weppmain/docs/readme.htm
Lawson, G., Burgess, P., Crowe, R., Mantzanas, K., Mayus, M., Moreno, G., McAdam, J. H., Newman, S., Pisanelli, A., Schuman, F., Sibbald, A. R., Sinclair, F. L., Thomas, T., and Waterhouse, A., 2004. Policy support for agroforestry in the European Union, 1st World Congress of Agroforestry - Book of Abstracts. Orlando, USA, University of Florida - Institute of Food and Agricultural Sciences, p. 189. Available at: http://conference.ifas.ufl.edu/WCA/
Lawson, G., Dupraz, C., Liagre, F., Moreno, G., Paris, P., and Papanastasis, V., 2005. Options for Agroforestry Policy in the European Union. Deliverable 9.3. SAFE - EU Research Project contract QLK5-CT-2001-00560, 34 pp. Available at: http://www.montpellier.inra.fr/safe/
Lehmann, J., and Schroth, G., 2003. Nutrient Leaching. In: Schroth, G., and Sinclair, F. (Eds.), Trees, Crops and Soil Fertility, CABI Publishing, Wallingford, pp. 151-166.
Lively, S. J., Gregory, P. J., and Buresh, R. J., 2000. Competition in tree row agroforestry systems I. Distribution and dynamics of fine root length and biomass. Plant and Soil. 227, 149-161.
MAPYA, 1999. SIG Oleícola Español, MAPYA - Ministerio de Agricultura, Pesca y Alimentación, Madrid. Available at: http://w3.mapya.es/dinatierra_v3/
McDonald, M., Lawrence, A., and Shrestha, P., 2003. Soil Erosion. In: Schroth, G., and Sinclair, F. (Eds.), Trees, Crops and Soil Fertility, CABI Publishing, Wallingford, pp. 325-343.
McKenney, D., Yemshanov, D., Fox, G., and Ramlal, E., 2004. Cost estimates for carbon sequestration from fast growing poplar plantations in Canada. Forest Policy and Economics. 6, 345-358.
Mermut, A. R., and Eswaran, H., 2001. Some major developments in soil science since the mid-1960s. Geoderma. 100, 403-426.

Metzger, M., Bunce, R., Jongman, R., Mücher, S., and Watkins, J. W., 2005. A climatic stratification of the environment of Europe. Global Ecology and Biogeography. 14, 549-563.
Middleton, H., 2001. Agroforestry and its effects on ecological guilds and arthropod diversity. MSc Thesis. Faculty of Forestry. University of Toronto, Ontario.
Mobbs, D., Lawson, G., and Brown, T., 2001. HyPARv4.1 Model for Agroforestry Systems User Guide. Centre for Ecology and Hydrology, Edinburgh. Available at: www.nbu.ac.uk/hypar/
Montagnini, F., and Nair, P. K. R., 2004. Carbon sequestration: An underexploited environmental benefit of agroforestry systems. Agroforestry Systems. 61, 281-295.
Morgan, R. P. C., 1995. Soil Erosion and Conservation. Longman, Harlow, UK, 198 pp.
Mücher, C. A., 2000. Development of a consistent methodology to derive land cover information on a European scale from remote sensing for environmental modelling. PELCOM project. Final report. EU Contract No ENV4-CT96-0315., 299 pp.
Nair, V. D., and Graetz, D. A., 2004. Agroforestry as an approach to minimizing nutrient loss from heavily fertilized soils: The Florida experience. Agroforestry Systems. 61, 269-279.
Nemeth, T., 1996. Nitrogen balances in long-term field experiments. Fertilizer Research. 43, 13-19.
Olea, L., and Figuera, F., 1999. Dehesa ecosystem: Production and preservation. In: M., E. editor, Dynamics and sustainability of Mediterranean pastoral systems, CIHEAM-IAMZ, Zaragoza, pp. 239-246.
Palma, J., Herzog, F., Reisner, Y., Graves, A., Burgess, P., Keesman, K., van Keulen, H., Mayus, M., De Filippi, R., and Bunce, R., 2005. Methodological approach for the assessment of environmental effects of agroforestry at the landscape scale. Ecological Engineering. Submitted.
-, 2006. Methodological approach for the assessment of environmental effects of agroforestry at the landscape scale. Ecological Engineering. Accepted.
Palma, J. H. N., and Reisner, Y., 2004. Working visit report on upscaling for seven landscape test sites in France. Agroscope FAL Reckenholz. Unpublished, Zurich, 15 pp.
Peng, R. K., Incoll, L. D., Sutton, S. L., Wright, C., and Chadwick, A., 1993. Diversity Of Airborne Arthropods In A Silvoarable Agroforestry System. Journal Of Applied Ecology. 30, 551-562.
Poulton, R., 1995. The importance of long-term trials in understanding sustainable farming systems: the Rothamsted experience. Australian Journal of Experimental Agriculture. 35, 825-834.
Reisner, Y., 2004. Working visit report on upscaling for three landscape test sites in The Netherlands. Agroscope FAL Reckenholz. Unpublished, Zurich, 9 pp.
Reisner, Y., De Filippi, R., Palma, J., and Herzog, F., 2006. Target regions for silvoarable agroforestry in Europe. Ecological Engineering. Accepted.
Reisner, Y., and Freyer, B., 2005. Modelling a resource conserving agricultural land use at regional scale. Agriculture, Ecosystems and Environment. submitted.
Renard, K., Foster, G., Weesies, G., McCool, D., and Yoder, D., 1997. Predicting Soil Erosion by Water: A Guide to Conservation Planning with the Revised Universal Soil Loss Equation (RUSLE), v. Washington, D.C. US Department of Agriculture, USDA Agricultural Handbook No.703
Richards, J. A., 1986. Remote Sensing Digital Image Analysis. An Introduction. Springer - Verlag, Berlin-Heidelberg-New York
Rowe, E. C., van Noordwijk, M., Suprayogo, D., Hairiah, K., Giller, K. E., and Cadisch, G., 2001. Root distributions partially explain N-15 uptake patterns in Gliricidia and Peltophorum hedgerow intercropping systems. Plant and Soil. 235, 167-179.
Sadras, V., 2002. Interaction between rainfall and nitrogen fertilisation of wheat in environments prone to terminal drought: economic and environmental risk analysis. Field Crops Research. 77, 201-215.
Sanchez, P. A., 1995. Science in Agroforestry. Agroforestry Systems. 30, 5-55.
Schröder, J., 1998. Towards Improved Nitrogen Management in Silage Maize Production on Sandy Soils. PhD Thesis. Wageningen Agricultural University, Wageningen.
Seligman, N., van Keulen, H., and Spitters, C., 1992. Weather, soil conditions and the interannual variability of herbage production and nutrient uptake on annual Mediterranean pastures. Agricultural and Forest Meteorology. 57, 265-279.

Seobi, T., Anderson, S. H., Udawatta, S. H. P., and Gantzer, C. J., 2005. Influence of Grass and Agroforestry Buffer Strips on Soil Hydraulic Properties for an Albaqualf. Soil Science Society of America Journal. 69, 893-901.

Sidle, R. C., Ziegler, A. D., Negishi, J. N., Nik, A. R., Siew, R., and Turkelboom, F., 2006. Erosion processes in steep terrain - Truths, myths, and uncertainties related to forest management in Southeast Asia. Forest Ecology And Management. 224, 199-225.

Smart, S. M., Bunce, R. G. H., Firbank, L. G., and Coward, P., 2002. Do field boundaries act as refugia for grassland plant species diversity in intensively managed agricultural landscapes in Britain? Agriculture, Ecosystems & Environment. 91, 73-87.

Stoate, C., Araújo, M., and Borralho, R., 2003. Conservation of European farmland birds: abundance and species diversity. Ornis Hung. 12-13, 33-41.

Thevathasan, N. V., and Gordon, A. M., 2004. Ecology of tree intercropping systems in the North temperate region: Experiences from southern Ontario, Canada. Agroforestry Systems. 61, 257-268.

Udawatta, R. P., Krstansky, J. J., Henderson, G. S., and Garrett, H. E., 2002. Agroforestry practices, runoff, and nutrient loss: A paired watershed comparison. Journal of Environmental Quality. 31, 1214-1225.

van der Knijff, J. M., Jones, R. J. A., and Montanarella, L., 2000. Soil erosion Risk Assessment in Europe. European Soil Bureau, Space Applications Institute. European Commission. European Communities., Brussels

van der Werf, W., Keesman, K., Burgess, P., Graves, A., Pilbeam, D., Incoll, L., Metselaar, K., Mayus, M., Stappers, R., van Keulen, H., Palma, J., and Dupraz, C., 2006. Yield-SAFE: a parameter-sparse process-based dynamic model for predicting resource capture, growth and production in agroforestry systems. Ecological Engineering. Accepted.

van Genuchten, M. T., 1980. A closed -form equation for predicting the hydraulic conductivity of unsaturated soils. Soil Science Society of America Journal. 44, 892-898.

van Ittersum, M., and Rabbinge, R., 1997. Concepts in production ecology for analysis and quantification of agricultural input-output combinations. Field Crops Research. 52, 197-208.

van Keulen, H., 1982. Graphical analysis of annual crop response to fertilizer application. Agricultural Systems. 9, 113-126.

van Kooten, G., Shaikh, S., and Suchánek, P., 2002. Mitigating Climate Change by planting trees: The transaction costs trap. Land Economics. 78, 559-572.

van Kooten, G. C., 2000. Economic Dynamics of Tree Planting for Carbon Uptake on Marginal Agricultural Lands. Canadian Journal of Agricultural Economics. 48, 51-65.

van Noordwijk, M., Lawson, G., Soumaré, A., Groot, F., and Hairiah, K., 1996. Root distribution of trees and crops: Competition and/or complementary? In: Huxley, P. editor, Tree-crop interaction: A physiological approach, University Press, Cambridge, pp. 319-364.

van Remortel, R., Hamilton, M., and Hickey, R., 2001. Estimating the LS Factor For RUSLE Through Iterative Slope Length Processing of Digital Elevation Data Within ArcInfo GRID. Cartography. 30, 27-35.

Webster, C. P., Conway, J. S., Crew, A. P., and Goulding, K. W. T., 2003. Nitrogen leaching losses under a less intensive farming and environment (LIFE) integrated system. Soil Use and Management. 19, 36-44.

Whitehead, D. C., 1995. Nitrogen Grassland. CAB International, Wallingford, 416 pp.

Wösten, J., Lilly, A., Nemes, A., and Le Bas, C., 1999. Development and use of a database of hydraulic properties of European soils. Geoderma. 90, 169-185.

Young, A., 1989. Agroforestry for Soil Conservation. CAB International, Wallingford, UK