AUTHOR QUERY FORM

ELSEVIER	**Journal:** ECOENG **Article Number:** 1634	**Please e-mail or fax your responses and any corrections to:** **E-mail:** corrections.esil@elsevier.thomsondigital.com **Fax:** +353 6170 9272

Dear Author,

Any queries or remarks that have arisen during the processing of your manuscript are listed below and highlighted by flags in the proof. Please check your proof carefully and mark all corrections at the appropriate place in the proof (e.g., by using on-screen annotation in the PDF file) or compile them in a separate list.

For correction or revision of any artwork, please consult http://www.elsevier.com/artworkinstructions.

Articles in Special Issues: Please ensure that the words 'this issue' are added (in the list and text) to any references to other articles in this Special Issue.

Uncited references: References that occur in the reference list but not in the text – please position each reference in the text or delete it from the list.

Missing references: References listed below were noted in the text but are missing from the reference list – please make the list complete or remove the references from the text.

Location in article	Query / remark Please insert your reply or correction at the corresponding line in the proof
Q1	Please check corresponding author and fax number.
Q2	Please check the unit in '3.7 mg ha-1 year-1'.
Q3	Uncited references: This section comprises references that occur in the reference list but not in the body of the text. Please position each reference in the text or, alternatively, delete it. Any reference not dealt with will be retained in this section. Thank you.

Electronic file usage

Sometimes we are unable to process the electronic file of your article and/or artwork. If this is the case, we have proceeded by:

☐ Scanning (parts of) your article ☐ Rekeying (parts of) your article ☐ Scanning the artwork

Thank you for your assistance.

Ecological Engineering xxx (2010) xxx–xxx

Contents lists available at ScienceDirect

Ecological Engineering

journal homepage: www.elsevier.com/locate/ecoleng

Carbon allocation dynamics one decade after afforestation with *Pinus radiata* D. Don and *Betula alba* L. under two stand densities in NW Spain

E. Fernández-Núñez, A. Rigueiro-Rodríguez, M.R. Mosquera-Losada *

Crop Production Department, Escuela Politécnica Superior, Campus de Lugo, University of Santiago de Compostela, 27002 Lugo, Spain

ARTICLE INFO

Article history:
Received 9 September 2009
Received in revised form 19 January 2010
Accepted 21 March 2010
Available online xxx

Keywords:
Silvopastoral systems
Pine
Birch
Carbon
Production
Growth

ABSTRACT

Silvopastoral systems can contribute to the mitigation of climate change by functioning as sinks for greenhouse gases better than exclusively agricultural systems. Tree species, density, and an adequate management of the pasture carrying capacity contribute to the capacity of carbon sequestration. In this study, the capacities for carbon sequestration in silvopastoral systems that were established with two different forest species (*Pinus radiata* D. Don and *Betula alba* L.) and at two distinct densities (833 and 2500 trees ha^{-1}) were evaluated. Tree, litterfall, pasture and soil carbon storage determinations were carried out to deliver carbon sequestration in the different pools within the first 11 years of a plantation establishment. The results show that the global capacity for carbon sequestration in silvopastoral systems with pine canopy was higher than with birch cover. Independently of the forest species, the capacity for carbon sequestration increased when the systems were established at higher plantation densities. There were found strong differences in the relative proportions of carbon in each component of the system (litterfall, tree, pasture and soil). The soil component was found to be most important in the case of the broadleaf forest established at low density. The establishment of a silvopastoral system enhanced soil carbon storage, since afforestation was carried out, which results in a more enduring storage capacity compared with treeless areas.

1. Introduction

Carbon sequestration by forests is an important environmental issue since the Kyoto Protocol (article 3.3) was adopted in 1997 (http://unfccc.int/resource/docs/convkp/kpeng.pdf). That resolution included the removals by sinks that result directly from human-induced land use changes and forestry activities to meet the Kyoto carbon emissions commitments by the involved countries in the determined periods from 1990 onwards (Mosquera-Losada et al., 2009). These facts make reforestation and afforestation, as well as deforestation, very important for the global carbon balance accounting of different countries. Reforestation of agricultural land will not only contribute to an increase in carbon sequestration on a global scale; it will also increase the supply of lumber, reducing the need for the logging of old-growth forests that, consequently, releases high amounts of stored carbon (Nair et al., 2008).

To verify compliance with the Kyoto Protocol, it is vital to measure the carbon sequestration caused by land use changes from agricultural to forestland, as well as the management of these lands. Reforestation of agricultural land has recently been promoted in Europe and has resulted in the reforestation of more than one million hectares throughout Europe between 1994 and 1999 (EC, 2005), a result of the implementation of Regulation No. 2082/92 (EU, 1992). The establishment of agroforestry in forestlands were promoted through direct payments in the last European Union Rural Development Council Regulation 1698/2005 (EU, 2005), making it necessary to evaluate the gains and losses of carbon caused by changes in tree biomass, pasture production, soil organic matter content and livestock greenhouse carbon (GHC) emissions. This also highlights the importance of evaluating the balance of different alternatives of forest management in different environments, as described by Gordon et al. (2005).

Forest carbon stocks are affected by the previous land use, tree species, tree density and the interaction of all these variables with climate (Reynolds et al., 2007).). In an agroforestry system, edaphic carbon is considered the most important store from a quantitative perspective (Dixon, 1995). The capacity to increase the sequestration of carbon in the soil will largely depend on the tree species used in reforestation and their density. Carbon storage in a silvopastoral system is balanced by the

Q1 * Corresponding author. Tel.: +34 600942437; fax: +34 982285926.
E-mail address: mrosa.mosquera.losada@usc.es (M.R. Mosquera-Losada).

0925-8574/$ – see front matter © 2010 Published by Elsevier B.V.
doi:10.1016/j.ecoleng.2010.03.007

emissions of greenhouse gases (CH_4 and N_2O) produced by the ruminants that feed on it. The amount of greenhouse gases, here called GHG emitted by livestock depends on the stocking rate, which depends on pasture production that is affected by tree development after afforestation. Thus, these should also be evaluated.

Agroforestry systems are not broadly extended within the Atlantic area of the European Union, where the important growth of trees could improve the European union carbon sequestration. Carbon sequestration studies carried out in the Atlantic region of Europe are related to grasslands or crops but not to forestlands, where aspects related to above and belowground carbon sequestration should be evaluated. Moreover, comparisons between tree species development and densities and their effect on livestock GHG emissions as well as on carbon sequestration should be carried out as pasture production and the chemical composition, and the quantity and rate of incorporation of carbon to soil from litterfall depends on tree species identity and density (Prescott et al., 2000). Compared with exclusively forest systems, carbon in silvopastoral systems should be evaluated. Some estimates assert that livestock production accounts for 18% of climate change, produces 9% of CO_2 emissions, 37% of CH_4 emissions, and 65% of the N_2O (Steinfeld et al., 2006). Moreover, long term studies should be carried out to quantify global carbon sequestration as tree canopy in the Atlantic region is fast developed, which affects to the global system biomass production (tree, pasture and therefore livestock), the inputs of organic matter into the soil and therefore the global carbon sequestration in the different pools of the agroforestry systems.

This paper aims to evaluate the amount of carbon sequestration in two silvopastoral systems that were established at two densities of *Pinus radiata* D. Don (pine) or *Betula alba* L. (birch) during the 11 years after trees were planted.

2. Materials and methods

2.1. Characteristics of the study site

The experiment was conducted in Castro Riberas de Lea (province of Lugo, NW Spain) at a latitude of 43.01′N and a longitude of 7.40′W. The study area is situated 439 m above sea level. The experiment was conducted in soil classified as an Umbrisol (FAO, 1998) with a sandy-loam texture (61.14% sand, 33.79% silt, 5.07% clay) that was previously designated for agricultural use (potato cultivation). The soil has an A horizon of 32 cm in depth, with some parts exceeding 40 cm. Argilic horizons began at a mean depth of 58 cm. According to the soil FAO classification system these soils are Umbrisol, with some horizon development, the eluviation of clay-sized particles to deeper horizons. These acidic and seasonally wet soils do not have accumulations of inorganic carbonates. The initial water pH (1:2.5) was nearly neutral (6.8), indicating to us a good availability of nutrients for plants (Porta-Casanellas et al., 2003). The edaphic contents of organic matter and nitrogen were 8.03% and 0.33%, respectively. Therefore, these would be considered elevated, though this is characteristic of soils used for cultivation in Galicia (Calvo de Anta et al., 1992). Furthermore the soil C/N ratio was 14.11, indicating a slow mineralisation rate and, consequently, favouring soil organic matter accumulation. The zone in which the experiment was conducted corresponds to what is considered an Atlantic bioclimatic region (EEA, 2003). The annual precipitation and the annual average temperature over the last 30 years were 1300 mm and 12.2 °C, respectively. Generally, moisture deficits that limit vegetative growth have been recorded in July and August due to drought.

2.2. Establishment, experimental design, and management

The experiment was initiated in 1995 and the results of the study were obtained for the period between 1995 and 2005. At the end of the winter of 1995, land ploughing was carried out. The results reported in this article pertain to a study involving 24 treatments. Some of the results have been previously reported in other publications (Rigueiro-Rodríguez et al., 2000; Mosquera-Losada et al., 2006; Fernández-Núñez et al., 2007). This article examines the results obtained for 4 of the treatments and 3 replicates (12 experimental units) that represent the typical forest management practices used in this area. The experimental design was random blocks with three replicates for each tree density. The treatments consisted of the evaluation of *P. radiata* (transplanted in soil from paperpots) and *B. alba* (bare rooted) that were established at two densities: (a) 2500 trees ha^{-1}, with a planting distance of 2 m × 2 m and an area of 64 m^2 per replicate, and (b) 833 trees ha^{-1}, with a planting distance of 3 m × 4 m and an area of 192 m^2 per replicate. In each experimental unit, 25 trees were planted with an arrangement 5 × 5 stems. After plantation, the plots were sown with a mixture of *Dactylis glomerata* L. var. Saborto (25 kg ha^{-1}) + *Trifolium repens* L. var. Ladino (4 kg ha^{-1}) + *Trifolium pratense* L. var. Marino (1 kg ha^{-1}). Fertiliser was not applied to replicate traditional reforestation practices for agricultural land in this area. A low pruning (at 2-m height) was performed on *P. radiata* at the end of 2001 and the *B. alba* was given a formational pruning with the objective of producing quality timber.

2.3. Field samplings

2.3.1. Soil

In order to determine the soil C content, a random sample was taken in January 2006 from each plot using a drill at a sampling depth of 25 cm, where the most organic matter accumulates. Once the samples were collected, they were taken to the laboratory, air-dried and sieved through a 2 mm screen. After this preparation, we determined the pH in water (1:2.5) and the total C content using the Saverlandt method (Guitián-Ojea and Carballás-Fernández, 1976).

2.3.2. Trees

Tree diameter measurements for *P. radiata* and *B. alba* were collected during the last year of the study (December 2005). The diameter of each inner plot tree was measured using a caliper at 1.30 m from the ground (diameter at breast height). Measurements were taken from nine inner trees in each plot. The biomass contents of the trees were determined via the implementation of allometric equations based on diameter (Table 1). These equations were determined by the National Institute of Agricultural Research and Technology and Food of Spain (Montero et al., 2005) in the region of the present study with tree densities similar to the experiment and have been used in the national carbon accounting system, as *P. radiata* stands are exclusively placed in the Atlantic Biogeographic Region of Spain, where the present study was developed.

2.3.3. Forest floor litter

The forest floor litter, hereafter litterfall, generated by the trees, which then accumulates on the soil surface, must be taken into account in estimates of a carbon cycle balance. The pine needle litterfall was hand separated from the same samples used for pasture production, as will be described in the next paragraph. No count was taken of the fallen birch leaves in the plot since the count of the birch leaves (being a deciduous species) was included in the estimate of the aboveground biomass of the tree (Table 1).

2.3.4. Pasture

Table 1
Values of the parameters *a* and *b* for the function $Y = e^{\mathrm{SEE}^2/2} \times e^a \times d^b$, the adjusted coefficient of determination (R^2), and the standard error of the estimation (SEE) for each of the species and each fraction of the biomass, where SEE: standard error of estimation; *d*: diameter (cm); BF: biomass of trunk; BR_7: biomass of branches with a diameter greater than 7 cm; BR_{2-7}: biomass of the branches with diameter between 2 and 7 cm; BR_2: biomass of the branches of diameter less than 2 cm; BA: needle biomass; BH: leaf biomass and Br: root biomass. (*Source*: Montero et al., 2005.)

Function	$Y = e^{\mathrm{SEE}^2/2} \times e^a \times d^b$			
Parameters				
Y	*a*	*b*	R_2^{adj}	SEE
Pinus radiata D. Don				
BF	3.02878	2.56358	0.976	0.20008
BR_7	10.5693	3.64861	0.710	0.52533
BR_{2-7}	4.12515	2.1173	0.746	0.61540
BR_2	3.53532	1.75877	0.669	0.61607
BA	5.03445	2.05803	0.739	0.60952
Br	2.78485	2.14449	0.939	0.30954
Betula spp.				
BF	2.09231	2.32560	0.970	0.161110
BR_7	7.84245	3.25429	0.476	0.683245
BR_{2-7}	2.70462	1.97187	0.871	0.297643
BR_2	2.65716	1.64983	0.747	0.373270
BH	3.28444	1.59452	0.720	0.386253
Br	2.41805	2.01124	0.775	0.402970

2.3.4.1. Aboveground biomass. During the 11 years studied, in each plot, the pasture was harvested using a hand harvester between six of the nine most central trees to avoid the border effect. Thus, areas of 24 m^2 and 8 m^2 were sampled for 833 and 2500 trees ha^{-1}, respectively. The samples were collected in May, June, July and December, as is traditional for the area, when the pastures reached about 20 cm. A sub-sample was taken, labelled and delivered to the laboratory. Once in the laboratory, two samples (100 g each) were taken to determine the relative proportions of the litterfall and pasture components after hand separation. These samples were oven-dried (72 h × 60 °C) to quantify the contribution (kg DM ha^{-1}) of litterfall and pasture components to the carbon sequestration model. From 2003 onwards, including the harvests from May and June of the same year, pasture biomass was no longer measured in those plots forested with pine at 2500 trees ha^{-1} because pasture production in these stands was nearly zero. The aboveground component of the pine system was comprised primarily of litterfall, since the tree canopies had become tangential. In these same plots, pasture production was estimated by harvesting sampling quadrats of 1 m × 1 m in July and December. Once sub-sampled, the remaining litterfall was not removed from the plot after 2003.

2.3.4.2. Belowground biomass. The carbon content of roots more than 2 mm in diameter was determined by the allometric relationships described in Table 1. To determine the carbon content in roots less than 2 mm in diameter (no distinction was made between tree and grass roots), samples were taken during the fall of the final year of the study at a depth of 15 cm (using a drill 5.1 cm in diameter). Samples were then sieved (with a 2-mm mesh screen) and pressure-washed with water. This sampling time was chosen because during this period, there are fewer living roots in the soil due to summer drought and the following precipitation that facilitates their incorporation into the soil. These values could represent a basal level that would increase in periods (e.g. spring) more conducive to the growth of the herbaceous component. Then, the samples were air-dried and the root:shoot ratio of the pasture was determined to estimate the root biomass present in the plots in 2005.

2.4. Carbon balance estimation

2.4.1. System description

To compare the carbon balance of the system, three main components were considered: tree, soil and pasture (including animal losses), as shown in Fig. 1. The C stored in trees and soil was estimated using data from 2005, while that of pasture was the average of samples collected between 1995 and 2005. With the goal of quantifying the potential GHG effect of the animals, we determined an average annual pasture carrying capacity (PCC) that the system could support based on actual annual pasture production in each treatment (Steinfeld et al., 2006).

When calculating the potential GHG effect of livestock from pasture production, the proportion of stable period/grazing period must be taken into account according to the habitual pasture production practices of the area (Mosquera and González, 1998), which are determined by the seasonal interaction of precipitation and temperature. Over the year, livestock is kept on pasture approximately 7 months (April, May, June, July, 15 days in September, October, November and 15 days in December) and stabled for the remaining 5 months, during which the animals feed on grass silage (approximately 150 days $year^{-1}$). The inclusion of the stabling period in the global calculation of C is very important because losses from the emission of N_2O from livestock occur only during this stabling period (IPCC, 1996). Of the various systems of management proposed for the sheep that are raised for meat production in Galicia (Zea-Salgueiro, 1992), those that are best adapted to the conditions in our system are for sheep of the Galician breed of 35 kg of live weight.

Silage area was taken into account to provide the same annual basis C measurements for tree (which were growing up all the year in the plots) and animals which were feed 210 days based on grazing and 150 days on silage.

In order to calculate the annual stocking rate we sum up the grazing area and the silage area. We deliver the number of animals to be fed during the grazing period by taking into account the real pasture production obtained under trees, afterwards, we calculate the kilos of silage needed by these animals and, later, the number of hectares needed to produce pasture to produce silage, and this area is used to estimate annual stocking rate.

2.4.2. Estimation of pasture carrying capacity (PCC)

From the data of annual pasture production (Mg DM ha^{-1}) and the forage necessary for sheep livestock (1.74 kg DM $sheep^{-1}$ day^{-1}) in a pasture (Flores et al., 1992), we employed Eq. (1) to estimate the pasture carrying capacity (PCC).

$$\mathrm{PCC}\ (\mathrm{sheep\ ha^{-1}}) = \frac{\mathrm{P}}{\mathrm{C}} \qquad (1)$$

where PCC is the pasture carrying capacity; P is the annual pasture production; and C forage requirements of grazing sheep for 210 days.

From the silage needs of 0.75 kg DM $sheep^{-1}$ day^{-1}, as cited by Flores et al. (1992), the known PCC, and the number of full days per year that sheep are stabled (150 days), we determined the average silage requirements using Eq. (2).

$$\mathrm{Total\ need\ of\ silage} = 0.75 \times \mathrm{PCC} \times 150 \qquad (2)$$

To determine the area required for grass cultivation for silage, we considered a treeless area and used data that represent conditions typical to Galicia: 1 ha of grass produces 7096 Mg DM of silage per $year^{-1}$ (Mosquera and González, 1998). This production includes losses that occur in the harvesting process and those that result from the processing of silage as well. The pasture area needed to produce silage adequate to support PCC was then estimated with

Please cite this article in press as: Fernández-Núñez, E., et al., Carbon allocation dynamics one decade after afforestation with *Pinus radiata* D. Don and *Betula alba* L. under two stand densities in NW Spain. Ecol. Eng. (2010), doi:10.1016/j.ecoleng.2010.03.007

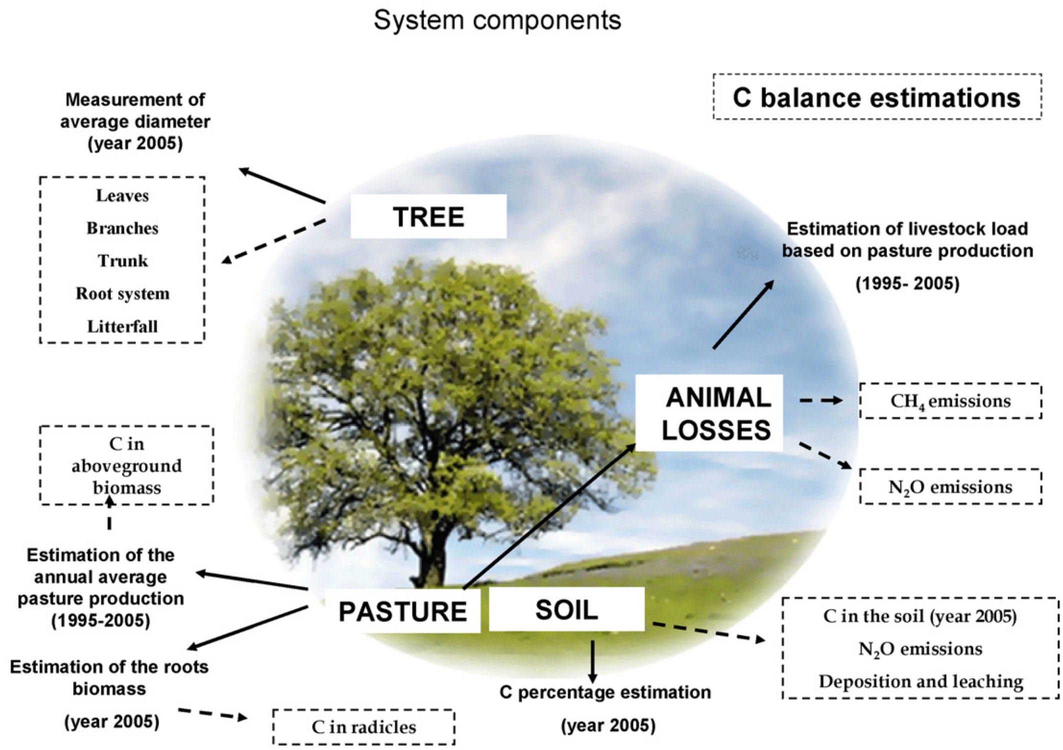

Fig. 1. Components of the system considered in order to evaluate the carbon balance in the study. The sampling period or year used to estimate the balance is shown between brackets.

Eq. (3).

$$\text{Silage area} = \frac{\text{silage needed}}{\text{silage production/ha}} \quad (3)$$

After determining the pasture area needed for silage production to feed the flock that would be supported on our silvopastoral system, we estimated the general system stocking rate (SR_{annual}). This metric captures the land area that is needed to maintain the livestock annually and is calculated using Eq. (4).

$$SR_{annual} = \frac{PCC}{\text{pasture area+silage area}} \quad (4)$$

The pasture area was 1 ha because the calculation used to determine the livestock sustained by pasture production in the silvopastoral system was 1 ha (Eq. (1)).

These figures were used to calculate the GHG emissions generated by the livestock for each year. The global carbon balance was determined using the average of those values.

2.4.3. Soil carbon estimation

2.4.3.1. Soil carbon storage. Once the actual percentage of edaphic carbon was estimated in the laboratory, the content of carbon in each of the treatments was calculated taking into account the soil density (1.1 Mg m^{-3}) and the sample depth via Eq. (5). It was found that soil density in the experiment did not significantly vary between tree species or densities (Howlett, 2009).

$$C\ (\text{Mg ha}^{-1}) = \frac{\%C \times \text{soil volume} \times \text{soil density}}{100} \quad (5)$$

As most of the C was already on the soil before the plantation, to estimate the C accumulated during those 11 years the difference between the C in 2005 and that already in the system in 1995 was calculated and divided by the years of the study (11).

2.4.3.2. Soil carbon losses. Following the Guidelines of the IPCC (1996), the direct and indirect N_2O emissions were calculated for the soil component in the different established systems (Fig. 2), which were derived from the pasture carrying capacity of the system previously calculated based on the actual pasture production. To determine the equivalent CO_2 amounts due to the N_2O emissions, the N_2O emissions were multiplied by the warming potential of N_2O, which corresponds to a value of 310 based on the IPCC report (1996).

(a) Direct emissions of edaphic N_2O

a.1 Stabling period

The direct emissions of N_2O resulting from the use of manure as fertiliser were determined (Fig. 2). To do so, we calculated the N excreted by the livestock (Nex) using the previously calculated animal stocking rate and we then determined the N in the manure used as fertiliser (Fe). Next, an adjustment was made to the NH_3 and NO_x emissions (Mosier et al., 1998; IPCC, 1996), excluding the manure produced during grazing.

a.2 Grazing period

Estimates of N_2O emissions during the grazing period were calculated by using the N excreted by the livestock (Nex), using the pasture carrying capacity previously calculated, and taking into account the emission factor established by the IPCC (1996) for this type of land use (Fig. 2).

(b) Indirect N_2O emissions

The emissions of NH_3 and NO_x resulting from the atmospheric deposition and those emissions due to leaching were calculated (IPCC, 1996). These emissions correspond to the N_2O that is indirectly produced from the N excreted by the livestock, and it was calculated by taking into account the pasture carrying capacity previously estimated. Through volatilization, a portion of this N enters into the atmosphere in the form of ammonia and oxides of nitrogen, and it later returns to the soil

Please cite this article in press as: Fernández-Núñez, E., et al., Carbon allocation dynamics one decade after afforestation with *Pinus radiata* D. Don and *Betula alba* L. under two stand densities in NW Spain. Ecol. Eng. (2010), doi:10.1016/j.ecoleng.2010.03.007

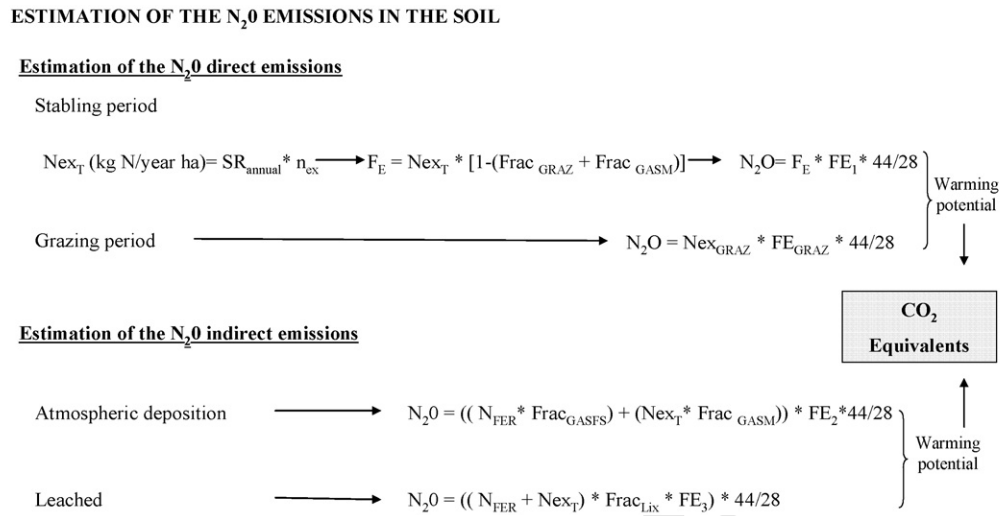

Fig. 2. Estimation of the N_2O emissions (CO_2 equivalents) from the livestock for the soil, where Nex_T = total N excreted by the livestock (grazing + stabled); SR_{annual} = stocking rate (sheep/ha); n_{ex} = nitrogen excreted from manure (20 kg N/animal unit/year (IPCC, 1996)); F_E = N input from manure (kg N ha year^{-1}); $Frac_{GRAZ}$ = Nex_T fraction during grazing; FE_{GRAZ} = emission factor (0.02 kg N_2O-N/kg N); $Frac_{GASM}$ = fraction of the total N excreted that is emitted as NO_x or NH_3 (kg N/kg N = 0.2 kg NH_3-N + NO_x-N/kg N); FE_1 = emission factor (0.0125 kg N_2O-N/kg of N input); N_{FER} = N applied in the fertilisation treatments (N_{FER} = 0); $Frac_{GASFS}$ = fraction of the N applied in the fertiliser that is volatilized (when fertiliser is not applied then $Frac_{GASFS}$ = 0); $Frac_{GASM}$ = fraction of the total N excreted that is volatilized (0.02 kg NH_3-N + NO_x/kg of N excreted by livestock); FE_2 = emission factor (0.01 kg N_2O-N per kg NH_3-N and NO_x-N emitted); $Frac_{LIX}$ = fraction of leached N (0.3 kg N/kg N in manure); and FE_3 = emission factor (0.025 kg N_2O-N per kg of N leaching and runoff). The values for the different factors in the formula are from the IPCC (1996) and correspond to the study area characteristics and the livestock considered (sheep).

by means of atmospheric deposition. This increases the production of N_2O. Another portion is lost from the soil through surface runoff and leaching, merging with surface and subterranean waters from which a proportion of this N is emitted as N_2O (Fig. 2).

2.4.4. Tree carbon estimation

Using the equation established by Montero et al. (2005) for *P. radiata* and *Betula* spp. (Table 1) and the data obtained from measuring the tree diameter at breast height, the aerial biomass of the following components of the tree cover were determined: trunk, thin and thick branches, leaves and roots (Eq. (6)).

$$Y = e^{\mathrm{SEE}^2/2} \times e^a \times d^b \qquad (6)$$

where Y is the biomass variable (biomass of trunk, biomass of the branches with a diameter greater than 7 cm, biomass of branches with a diameter within 2 cm and 7 cm, biomass of branches with a diameter less than 2 cm, leaf and root biomass) and d is the diameter at breast height (cm).

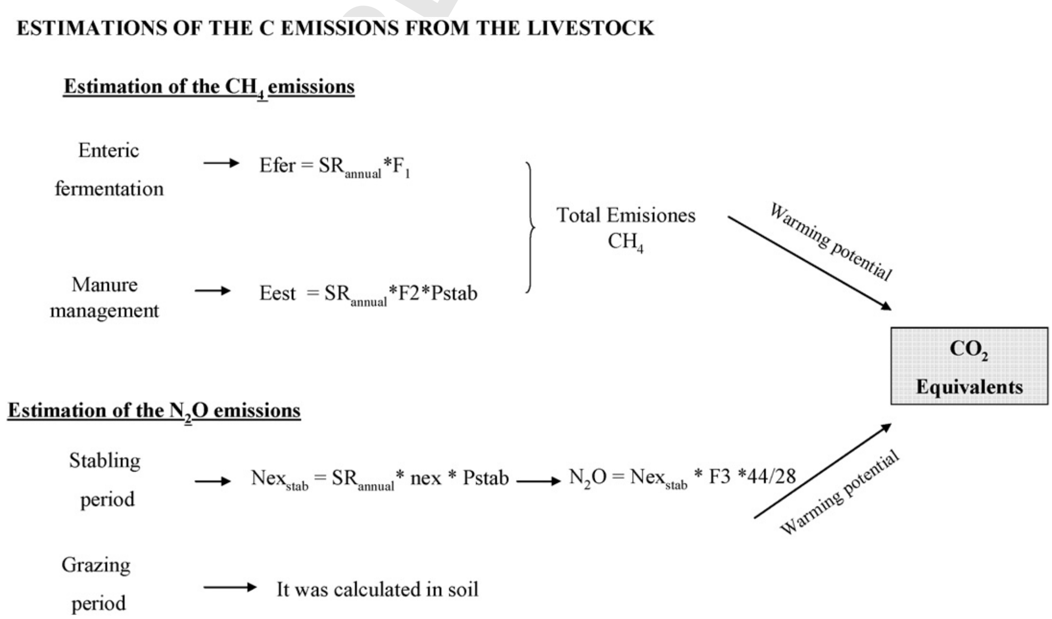

Fig. 3. Method used for the estimation of the CH_4 and N_2O emissions (CO_2 equivalents) from the livestock, where E_{fer} = emissions from the enteric fermentation; SR_{annual} = stocking rate (sheep/ha); F_1 = average emission factor (5 kg of CH_4/animal unit/year); E_{est} = emissions from manure management: F_2 = emission factor dependent on the monthly average temperature in the area (0.22 kg of CH_4/animal unit/year); P_{stab} = stabling period (150 days/year); Nex_{stab} = total N excreted by the livestock on stabling period); n_{ex} = excreted N (20 kg/animal unit/year); F_3 = emission factor for the stabling period (0.02 kg N_2O-N/kg N). The values for the different factors in the formula are from the IPCC (1996) and correspond to the study area characteristics and the livestock considered (sheep).

Please cite this article in press as: Fernández-Núñez, E., et al., Carbon allocation dynamics one decade after afforestation with *Pinus radiata* D. Don and *Betula alba* L. under two stand densities in NW Spain. Ecol. Eng. (2010), doi:10.1016/j.ecoleng.2010.03.007

Table 2
The pH and %C in soil (mean ± standard error) at the time of establishment of the system and in the years 2000 and 2005 under *Pinus radiata* and *Betula alba* at two densities (2500 and 833 trees ha^{-1}). Different letters indicate significant differences between treatments. ns: no significant difference.

Soil parameters	Year	Sig	2500 trees ha^{-1}		833 trees ha^{-1}	
			Pinus radiata	*Betula alba*	*Pinus radiata*	*Betula alba*
pH	Initial		6.8			
	2000	ns	5.4 ± 0.12	5.4 ± 0.15	5.1 ± 0.00	5.8 ± 0.00
	2005	*	5.7 ± 0.13b	5.9 ± 0.14ab	6.3 ± 0.32a	6.2 ± 0.11a
%C	Initial		4.6			
	2000	**	5.84 ± 0.45a	6.02 ± 0.16a	4.39 ± 0.54b	4.10 ± 0.42b
	2005	ns	5.30 ± 0.99	6.22 ± 0.67	4.75 ± 0.35	5.21 ± 0.15
Mg C ha^{-1}	Initial		126.50			
	2000	**	160.74 ± 12.45a	165.68 ± 4.54a	120.68 ± 14.91b	112.86 ± 11.72b
	2005	ns	145.80 ± 27.21	171.00 ± 18.34	130.69 ± 9.68	143.41 ± 4.30

* $P < 0.07$ for pH.
** $P < 0.05$ for C and Mg C ha^{-1}.

In Table 1, the values of SEE are shown, as well as the parameters *a* and *b* that were applied to calculate the biomass of each fraction. Once this value was obtained, the C content for this biomass was calculated by multiplying by an average value of 0.50 (Merino et al., 2003; Montero et al., 2005).

2.4.5. Litterfall

The litterfall C content in the last year was obtained by multiplying the litterfall biomass (Mg DM ha^{-1}) by a factor of 0.49 (Gómez-Rey and Calvo de Anta, 2002).

2.4.6. Pasture carbon estimation

From the data of pasture production (Mg DM ha^{-1}) obtained for each of the treatments and in each year, the C content of the herbaceous stratum was determined, distinguishing between aboveground and belowground parts.

2.4.6.1. Aboveground. Above pasture C content could be divided in two fractions: above and below 5 cm of aboveground pasture height. C content determination of the 5 first cm of the pasture aboveground fraction was not included in the model because this is one of the sources of soil C, so it is already included in the system. However, C content determination of aboveground pasture placed above 5 cm from the soil was included because it will be mostly storaged in the animal bodies, once excluding livestock GHG emissions, on an annual basis. The C content corresponding to the aerial section of the herbaceous stratum was calculated on a yearly basis, based upon the pasture production during the pasture season and the need for silage. It was taken into account that when grass is converted to silage, it suffers a 15% loss in weight (Mosquera and González, 1998). Once the annual silage needed for the stabling period was estimated with actual data obtained from the pasture production that was attributed to the pasture season, we were able to quantify the organic matter content in it (Eq. (7)). The percentage of organic matter found in the pasture in Galicia is around 90.36% (Flores et al., 1992), and the C content in a pasture will be 50% of the organic matter (Montero et al., 2005).

$$\text{OM pasture} = (\text{Mg DM pasture ha}^{-1}) \times 0.9036 \tag{7}$$

2.4.6.2. Belowground. From the soil samples, and as in the procedure previously explained, we obtained a value of the ratio of root/aboveground biomass in the pasture that was 32.37%. Then, the root biomass was determined by applying this ratio to pasture production (pasture production during pasture season + pasture production during the stabling period). Once the root biomass was determined, the C content was estimated to be 49.67% of that value (Gordon et al., 2005).

2.4.7. Livestock

2.4.7.1. Estimation of livestock carbon losses. We estimated the CH_4 and N_2O emissions resulting from sheep livestock management, as well as their equivalents in terms of CO_2. The method used to estimate this emission is described by the IPCC (1996). In Fig. 3, the equation and coefficients used in this study are shown, again, following the protocol of the IPCC (1996) and the guidelines indicated for the regional estimation of carbon emissions established by the government of the region in which the study was conducted (Xunta de Galicia, 2004).

Table 3
Estimates of total N_2O emission (direct and indirect) in Mg ha^{-1} from the soil during the 11 study years under *Pinus radiata* and *Betula alba* at the 2 stand densities (2500 and 833 trees ha^{-1}).

Years 1995–2005	2500 trees ha^{-1}		833 trees ha^{-1}	
	Pinus radiata	*Betula alba*	*Pinus radiata*	*Betula alba*
Direct				
Stabling	5.44×10^{-3}	6.10×10^{-3}	5.85×10^{-3}	7.66×10^{-3}
Pasturing	24.42×10^{-3}	27.38×10^{-3}	26.27×10^{-3}	34.41×10^{-3}
Total	29.86×10^{-3}	33.48×10^{-3}	32.12×10^{-3}	42.07×10^{-3}
Equiv CO_2	9.25	10.38	9.96	13.04
Indirect				
Deposition	0.43×10^{-3}	0.47×10^{-3}	0.46×10^{-3}	0.59×10^{-3}
Leaching	15.54×10^{-3}	17.42×10^{-3}	16.72×10^{-3}	21.88×10^{-3}
Total	15.97×10^{-3}	17.89×10^{-3}	17.18×10^{-3}	22.47×10^{-3}
Equiv CO_2	4.95	5.54	5.32	6.96
Total Equiv CO_2	14.20	15.92	15.28	20.00
Equiv CO_2 $year^{-1}$	1.29	1.45	1.39	1.82

Table 4
Tree measurements of *Pinus radiata* and *Betula alba* (mean ± standard error) at two densities (2500 and 833 trees ha^{-1}) and site index estimation at 20 years (Is). Different letters indicate significant differences between treatments ($P < 0.001$).

Year 2005	2500 trees ha^{-1}		833 trees ha^{-1}	
Tree parameters	*Pinus radiata*	*Betula alba*	*Pinus radiata*	*Betula alba*
Basal diameter (cm)	15.0 ± 0.49a	6.2 ± 0.28b	16.8 ± 0.87a	7.2 ± 0.35b

(a) Estimated CH_4 emissions

a.1 Enteric fermentation estimates

To estimate enteric fermentation emissions, the pasture carrying capacity (CG annual) and the average emissions of CH_4 per animal per year were taken into account. In our case, with sheep, the value of the emission factor is 5 kg CH_4 $sheep^{-1}$ $year^{-1}$ (IPCC, 1996; Xunta de Galicia, 2004).

a.2 Manure management emissions

To estimate manure management emissions, only the 5 months of stabling were taken into account. The animals were stabled 41% of the days; thus, we multiply obtained values by 0.41, which is the distribution percentage of the use frequency in this type of manure management system. Following the IPCC methodology, once the CH_4 emissions from the livestock were obtained, the equivalent CO_2 was determined taking into account the warming potential of CH_4, which has been established as 21 by the IPCC (1996).

(b) Estimated N_2O emissions from livestock

The N_2O emissions result from both the stable and the pasture periods (IPCC, 1996).

b.1 Stabling period

Emissions of N_2O were calculated using the pasture carrying capacity previously estimated for each treatment. The quantity of excreted nitrogen (Nex) that resulted from manure management was calculated by taking into account the percentage of full days that livestock were stabled throughout the year (41%). An emission factor was then applied to this amount, which varies according to type of livestock being considered, and is 20 kg $animal^{-1}$ $year^{-1}$ for sheep (IPCC, 1996). Finally the CO_2 equivalents were determined taking into account that the warming potential of N_2O is 310 (IPCC, 1996).

b.2 Pasturing period

This was calculated in the soil component (IPCC, 1996).

2.5. Statistical analyses

The pH, C in soil, tree diameter, tree height, and annual pasture production variables were analysed by a factorial ANOVA, using treatments and blocks as factors within each year. The significant differences between means were determined using the LSD test (SAS, 2001).

3. Results

3.1. Soil

During the course of the 11-year study, significant acidification of the soils occurred. This is typical in the area due to high rainfall and high levels of soil cation extraction from crops that bring acidity in Galician soils. No significant differences were found between treatments in relation to pH in the first 5 years of system production (Table 2). However, 11 years later, there was a tendency for a significant decrease in pH ($P < 0.07$), especially in the higher density plantations under pine species. On the other hand, the results show a significant ($P < 0.05$) effect of treatments on the C content in the soil after 5 years of system development. A significant increase in the soil C content ($P < 0.05$) occurred in those systems with the higher tree density (independent of species planted), an effect which had disappeared at the 11-year mark (Table 2). From the time that the system was established (126.50 Mg C ha^{-1}), independent of the forest species used, an increase in the soil C content was observed in 2005 over the level present at the time of plantation establishment. This level of increase was greater in plots that were established at higher tree densities (15% higher under pine and 35% higher under birch).

3.1.1. Estimates of N_2O emissions in the soil

For the 11 years of the study, the estimates of N_2O emissions for each of the different systems is shown in Table 3, as are the equivalents in CO_2 emissions to the atmosphere. The results reflect higher emission levels in those systems that were supporting a higher pasture carrying capacity, i.e. those established under birch cover, independent of the tree density.

3.2. Trees

The diameter reached by *P. radiata* during the last year of the study was significantly higher than that of *B. alba* (Table 4). In regard to diameter, the results show a similar tendency of tree density on the development of each of the two forest species. Higher tree densities favoured the lowest diameter, due to tree competition.

3.2.1. Tree carbon

In both forest species, the highest carbon accumulation occurred in the aerial component (Table 5). In the conifer, high densities

Table 5
Total carbon in the tree biomass (Mg C ha^{-1}) determined for the year 2005 by taking into account the average diameter obtained for *Pinus radiata* and *Betula alba* at the two stand densities considered, where BF: trunk biomass; $BR_{>7cm}$: biomass of branches greater than 7 cm; BR_{2-7cm}: biomass of branches with diameters between 2–7 cm; $BR_{<2cm}$: biomass of branches less than 2 cm; BH: needles biomass (in pine) or leaf biomass (in birch); Br: root biomass.

	Total carbon		C aerial biomass (Mg C ha^{-1})					Root biomass (Mg C ha^{-1})		
	Density	*d* (cm)	BF	$BR_{>7cm}$	BR_{2-7cm}	$BR_{<2cm}$	BH	Total aerial	Br	Total
Pinus radiata	2500	15.03	62.95	0.71	7.43	5.08	2.54	78.71	26.89	105.61
	833	16.78	27.88	0.36	3.13	2.06	1.06	34.49	11.35	45.84
Betula alba	2500	6.25	10.07	0.00	2.95	1.76	0.85	15.63	4.67	20.30
	833	7.57	5.36	0.00	1.47	0.82	0.39	8.04	2.29	10.33

Please cite this article in press as: Fernández-Núñez, E., et al., Carbon allocation dynamics one decade after afforestation with *Pinus radiata* D. Don and *Betula alba* L. under two stand densities in NW Spain. Ecol. Eng. (2010), doi:10.1016/j.ecoleng.2010.03.007

Table 6
Amount of carbon content in the aboveground part of the pasture (pasture + silage) in the established *Pinus radiata* systems for each year of the study (1995–2005), where PCC = pasture carrying capacity; SR_{annual} = system stocking rate. Grazing and stabling period lasted 210 and 150 days per year. Food sheep requirements per day were 1.74 kg of pasture and 0.75 kg of silage. Silage production was 7096 kg DM silo per year. Letters in the pasture production column indicates significant differences between treatments within the same year.

Year	Pasturing period		Stabling period					
	Pasture production	PCC	Silage requirements	Silage area	SR_{annual}	Total herbaceous		Average C
	kg DM ha^{-1}	sheep ha^{-1}	kg DM silage ha^{-1} year^{-1}	ha	sheep ha^{-1}	(Pasture + silage) (kg DM ha^{-1})	Mg C ha^{-1}	(Mg C ha^{-1} year^{-1})
2500 trees ha^{-1} *Pinus radiata*								
1995	3450	9	1013	0.14	8	4463	2.02	
1996	3770	10	1125	0.16	9	4895	2.21	
1997	1280b	3	338	0.05	3	1618	0.73	
1998	3230	9	1013	0.14	8	4243	1.92	
1999	3230	9	1013	0.14	8	4243	1.92	
2000	2720b	7	788	0.11	6	3508	1.58	1.46
2001	5720ab	16	1800	0.25	13	7520	3.40	
2002	530b	1	113	0.02	1	643	0.29	
2003	1070	3	338	0.05	3	1408	0.64	
2004	1290	4	450	0.06	4	1740	0.79	
2005	960b	3	338	0.05	3	1298	0.59	
833 trees ha^{-1} *Pinus radiata*								
1995	5200	14	1575	0.22	11	6775	3.06	
1996	3300	9	1013	0.14	8	4313	1.95	
1997	3590ab	7	788	0.11	6	4378	1.53	
1998	1640	4	450	0.06	4	2090	0.94	
1999	1850	5	563	0.08	5	2413	1.09	
2000	3280b	9	1013	0.14	8	4293	1.94	1.63
2001	5590ab	15	1688	0.24	12	7278	3.29	
2002	2390a	6	675	0.10	5	3065	1.38	
2003	2220	6	675	0.10	5	2895	1.31	
2004	1640	4	450	0.06	4	2090	0.95	
2005	970b	3	338	0.05	3	1308	0.59	
2500 trees ha^{-1} *Betula alba*								
1995	3430	9	1013	0.14	8	4443	2.01	
1996	2860	8	900	0.13	7	3760	1.70	
1997	2010ab	5	563	0.08	5	2573	1.16	
1998	3270	9	1013	0.14	8	4283	1.94	
1999	2700	7	788	0.11	6	3488	1.57	
2000	2050b	6	675	0.10	5	2725	1.23	1.67
2001	2910b	8	900	0.13	7	3810	1.72	
2002	1310ab	4	450	0.06	4	1760	0.79	
2003	2650	7	788	0.11	6	3438	1.55	
2004	4060	11	1238	0.17	9	5298	2.39	
2005	4110a	11	1238	0.17	9	5298	2.39	
833 trees ha^{-1} *Betula alba*								
1995	4870	13	1463	0.21	11	6333	2.86	
1996	3700	10	1125	0.16	9	4825	2.18	
1997	3870a	11	1238	0.17	9	5108	2.31	
1998	3090	8	900	0.13	7	3990	1.80	
1999	3110	8	900	0.13	7	4010	1.81	
2000	5330a	15	1688	0.24	12	7018	3.17	1.63
2001	7680a	21	2363	0.33	16	10043	4.54	
2002	2020a	6	675	0.10	5	2695	1.22	
2003	2750	7	788	0.11	6	3538	1.60	
2004	1990	5	563	0.08	5	2553	1.15	
2005	2440ab	7	788	0.11	6	3228	1.46	

increase C fixation per unit surface area around 43% with respect to the lower density plantations, and in the deciduous species, this increase was 51%. The average C accumulation during the 11-year period in the *P. radiata* stand was 9.86 Mg C ha^{-1} year^{-1} at a density of 2500 trees ha^{-1} and 4.35 Mg C ha^{-1} year^{-1} with 833 stems ha^{-1}. In the *B. alba* stand, it was 1.84 and 0.94 Mg C ha^{-1} year^{-1} for the densities of 2500 and 833 trees ha^{-1}, respectively. If we compare the effect of density on the two forest species, we see that at triple the density, the carbon content in the aerial component of the timber doubled; this increase was slightly higher in the pine.

Table 7
Amount of carbon content (Mg ha^{-1}) in the roots of the herbaceous component of the systems evaluated.

Year 2005	2500 trees ha^{-1}		833 trees ha^{-1}	
	Pinus radiata	*Betula alba*	*Pinus radiata*	*Betula alba*
Pasture + silage (kg DM ha^{-1})	1298	5298	1308	3228
Root (kg DM ha^{-1})	420	1715	423	1045
Mg C ha^{-1}	0.21	0.85	0.21	0.52

Please cite this article in press as: Fernández-Núñez, E., et al., Carbon allocation dynamics one decade after afforestation with *Pinus radiata* D. Don and *Betula alba* L. under two stand densities in NW Spain. Ecol. Eng. (2010), doi:10.1016/j.ecoleng.2010.03.007

Table 8
Estimates of the total emissions (Mg ha^{-1}) of methane (E_{CH_4}) and oxides of nitrogen (E_{N_2O}) due to the manure management of livestock during the period between 1995 and 2005, where E_{fer}: CH_4 emissions from enteric fermentation; E_{est}: CH_4 emissions from manure management; Nex: total N excreted by livestock during the 11 years of the study, and Equiv CO_2: CO_2 equivalents (Mg ha^{-1}).

Years 1995–2005	2500 trees ha^{-1}		833 trees ha^{-1}	
	Pinus radiata	*Betula alba*	*Pinus radiata*	*Betula alba*
E_{CH_4}				
E_{fer}	0.330	0.370	0.355	0.465
E_{est}	6.0×10^{-3}	6.7×10^{-3}	6.4×10^{-3}	8.4×10^{-3}
Total	0.336	0.377	0.361	0.473
Equiv CO_2 (Mg ha^{-1})	7.06	7.91	7.58	9.93
E_{N_2O}				
Nex	0.541	0.607	0.582	0.762
N_2O	17×10^{-3}	19×10^{-3}	18×10^{-3}	24×10^{-3}
Equiv CO_2 (Mg ha^{-1})	5.3	5.9	5.6	7.4
Total Equiv CO_2 (Mg ha^{-1})	12.36	13.81	13.19	17.33

3.3. *Litterfall*

Litterfall content in the pine plots in 2005 was 6.25 Mg ha^{-1} at a density of 2500 trees ha^{-1} and 4.26 Mg ha^{-1} at 833 trees ha^{-1}. This resulted in an average C content of 3.06 Mg C ha^{-1} year^{-1} and 2.09 Mg C ha^{-1} year^{-1} at the higher and lower densities, respectively. Generally, as occurs with the aboveground biomass, the capacity for needle accumulation in the soil incrementally increases with stand density, which is attributed to the earlier canopy closure in the higher density stands. Of the total fixed carbon in the tree stratum, the percentage of carbon accumulation accounted for by the fallen needles was 2.9% at a tree density of 2500 trees ha^{-1} and 4.5% at 833 trees ha^{-1}. This indicates that, at triple the density, the greater litterfall increased carbon storage by approximately 55%. Therefore, as was found with carbon storage in the living tree component, this C also doubled.

3.4. *Pasture*

3.4.1. *Aboveground*

The results show a significant effect of the applied treatments on pasture production in the years 1997, 2000, 2001, 2002, and 2005 (Table 6). The average production during the course of the study at 2500 and 833 trees ha^{-1}, respectively, was 2.5 and 2.8 Mg ha^{-1} year^{-1} in the pine systems and 3.8 and 3.7 mg ha^{-1} year^{-1} under birch. Furthermore, during the trial, the increasing light interception significantly reduced pasture production in the pine stands, whereas under the birch, pasture production was more dependent from other climate parameters. On the other hand, in 2001, as a result of the low pruning in the systems and an unusually rainy summer, an increase in pasture production was observed, independent of the tree density or forest species.

Table 6 shows the C content measured in the aboveground herbaceous layer (pasture during grazing season + pasture for silage) throughout 11 years (1995–2005). The amount of C accumulated during the 11 years of system growth resulted in an increase of 1.46 Mg C ha^{-1} and 1.63 Mg C ha^{-1} under pine cover at 2500 and 833 trees ha^{-1}, respectively. In the systems established under birch, the estimates were 1.67 Mg C ha^{-1} and 2.19 Mg C ha^{-1} for the lower and higher densities, respectively.

3.4.2. *Belowground*

In 2005, the estimated amount of C in the fine roots was 0.21 Mg C ha^{-1} under pine for both of the two plantation densities. Under birch, we obtained estimates of 0.85 Mg C ha^{-1} and 0.52 Mg C ha^{-1} at 2500 and 833 trees ha^{-1}, respectively (Table 7).

3.5. *Estimation of livestock carbon losses*

The estimate of the total CH_4 and N_2O emissions from the livestock, as well as the equivalents in CO_2, are reported in Table 8. The emissions of CH_4 and N_2O on the part of the livestock were greater in those systems that combined lower plantation densities with deciduous tree coverage (although they were always less than 10 Mg CO_2 ha^{-1}) because these systems supported higher animal stocking rates.

3.6. *Balance of carbon*

The final balance of the carbon cycle, calculated for the different systems studied, is shown in Figs. 4 and 5, and the relative proportion of each component (pasture, litterfall, animals, trees, and soil) is given in Fig. 6. If we compare the capacity for carbon sequestration at the end of the experiment (Mg C ha^{-1} year^{-1}) among the different components of the system, the tree shows the highest level of C fixation, followed by soil, and finally, by pasture (tree > soil > pasture). The exception occurs in the systems planted with birch at low density, in which the C stored in the soil component is higher than that of the tree (soil > tree > pasture) due to the lower rate of tree growth.

Our estimates show a tendency, though not significant, of a greater capacity to fix carbon in the systems with higher plantation density (8.19 Mg C ha^{-1} year^{-1} in 2500 trees ha^{-1} and 6.75 Mg C ha^{-1} year^{-1} in 833 trees ha^{-1}), especially in the case of the pine. On the other hand, if we compare the two forest species, the estimates reflect a clear tendency ($P < 0.05$) of a greater carbon sequestration capacity in the silvopastoral systems planted with pine (10.95 and 3.99 Mg C ha^{-1} year^{-1} for *Pinus* and *Betula*, respectively).

Fig. 6 shows the relative proportion of the various system components in relation to their carbon sequestration at the end of the experiment. The relative contribution of each component to the carbon balance at the end of the experiment changes within each system. In the case of the birch, the contribution of carbon in the soil to the total carbon in the system is greater than in the pine ($P = 0.05$; 44% compared to 15%). This becomes especially pronounced at the lower stand density. In contrast, the relative contribution of the tree component to the total system, excluding litterfall, is higher in the pine than in the birch (81% of the total for the pine compared to 45% of the birch; $P = 0.05$). In all cases, livestock emissions remained counterbalanced by the carbon accumulated in the pasture and in the litterfall.

Focusing exclusively on the relative proportion of carbon sequestration according to the different pool of storage in the system (tree, pasture + litterfall, soil) (Table 9), we observed that the

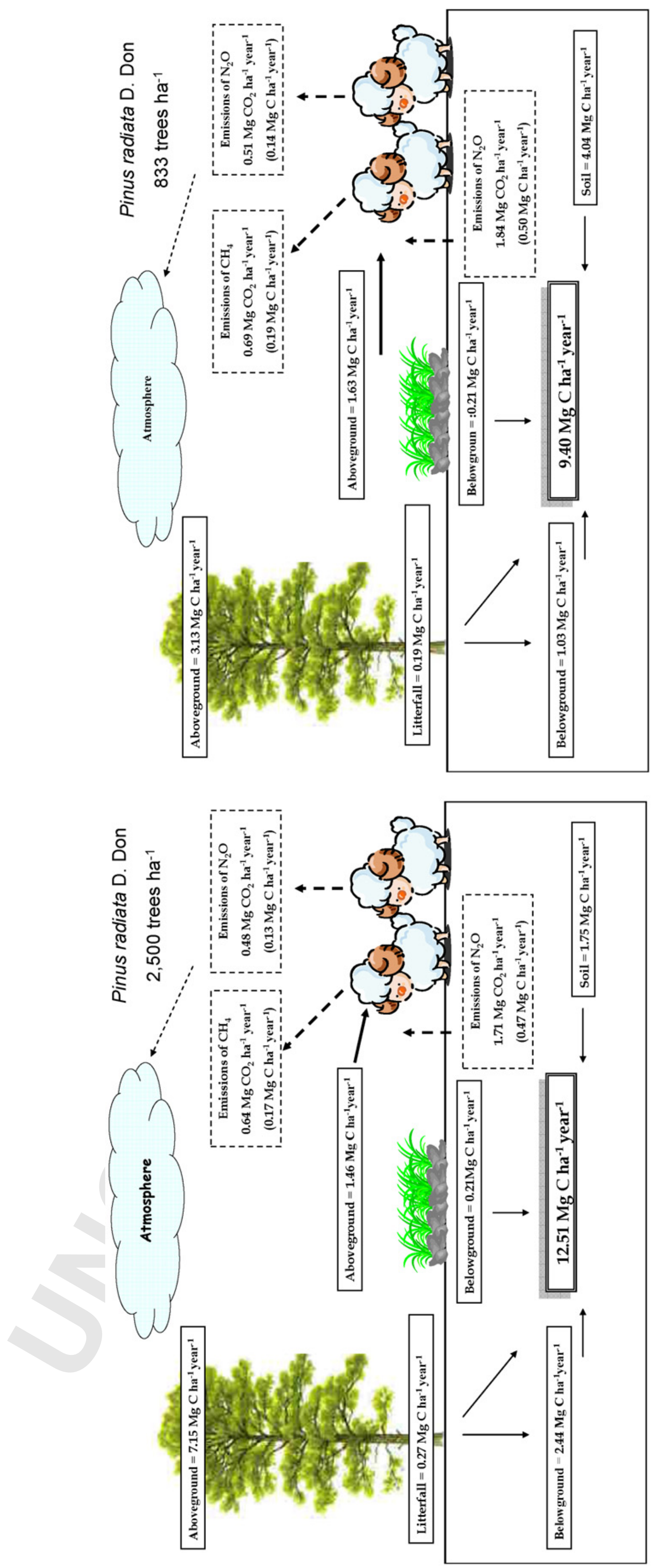

Fig. 4. Carbon balance (Mg C ha^{-1} year^{-1}) for those systems established under *Pinus radiata* D. Don at plantation densities of 2500 and 833 trees ha^{-1}.

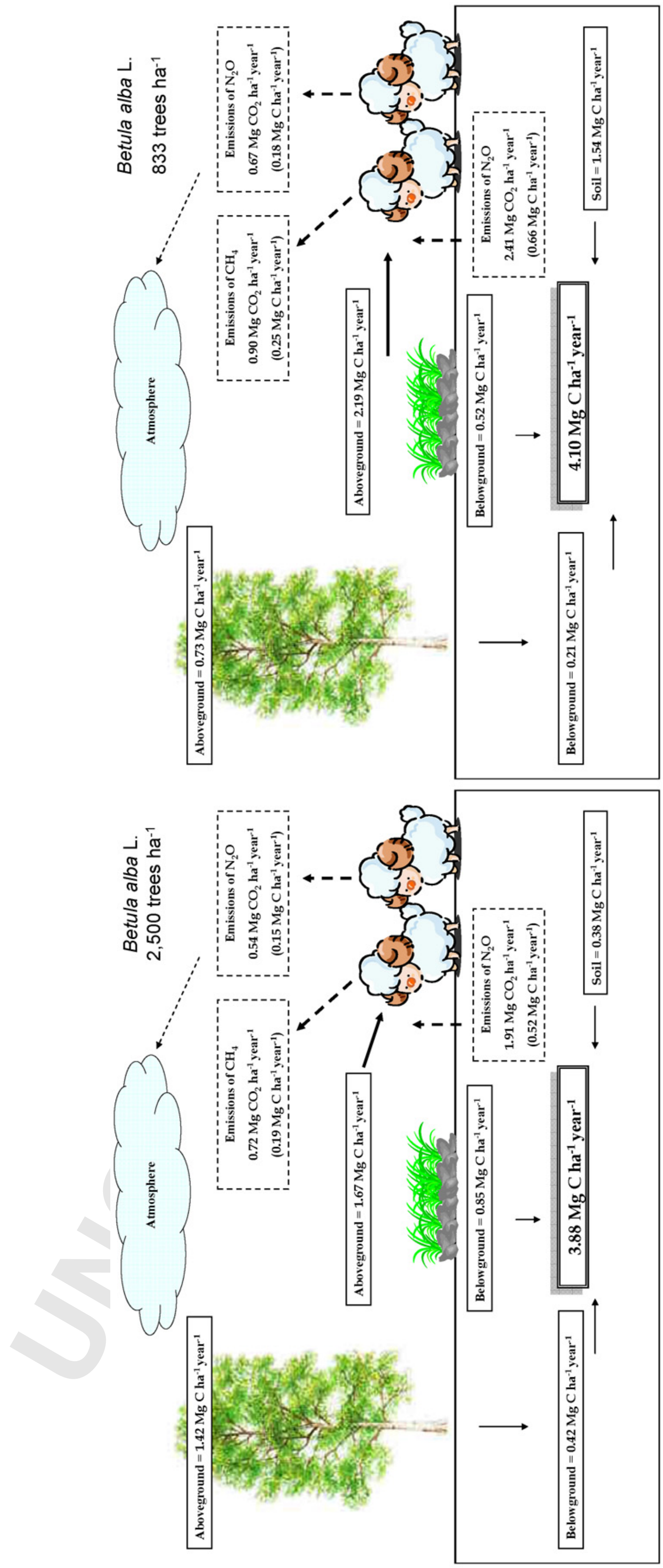

Fig. 5. Carbon balance (Mg C ha^{-1} year^{-1}) for those systems established under *Betula alba* L. at plantation densities of 2500 and 833 trees ha^{-1}.

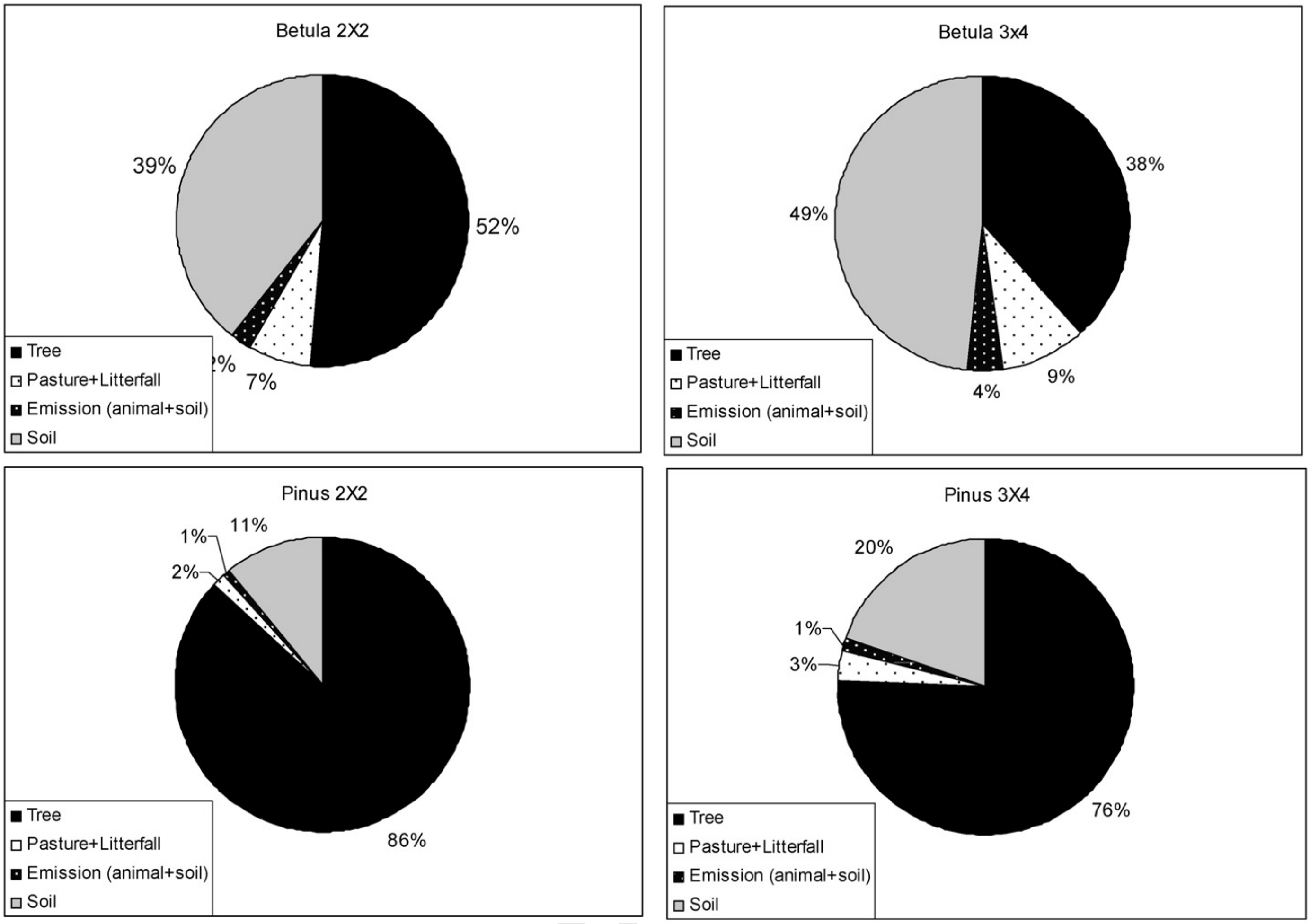

Fig. 6. Relative contribution to C storage in tree, pasture + tree litterfall, animal and soil components in each system expressed after 11 years of experiment.

storage differ markedly. The results show a far superior contribution of carbon from the trees from the more densely planted pines compared to all other treatments. Within the pine plantations, the stand density changes the percentage of C storage capacity of all of the components, as occurs in the birch system when the edaphic component is excluded. However, this contribution does not vary between forest species at the same plantation densities.

4. Discussion

Carbon sequestration in both soil and aboveground biomass is one of the most important benefits of the afforestation of agricultural lands (Maia et al., 2007; Nair et al., 2007). Carbon sequestration in woody biomass is promoted as a practice to offset increasing atmospheric CO_2 concentrations (Sauer et al., 2007). However, extensive analyses of forest productivity for various forest types and management practices have been primarily completed for tree aboveground biomass, usually without assessment of the understory.

The C content has been found to be higher in conifer forests due to the higher growth rate of this species compared to that of birch (Bunker et al., 2005; Kirby and Potvin, 2007). *P. radiata* has a greater C sequestration in the biomass compared to birch at the same plantation densities, by 21% at 2500 trees ha^{-1} and by 9.5% at 833 trees ha^{-1}, after 11 years.

The rate of carbon sequestration of the conifer plantations in our study was less than that normally expected in silvopastoral systems, like those of New Zealand, due not only to the lower tree density (Chang and Mead, 2003), but also to the higher site index

Table 9
Relative carbon allocation (estimated as Mg C ha^{-1}) to the storage pools of the different agroforestry systems. Different letters indicate significant differences between treatments in each of the components ($P < 0.05$).

	% Carbon allocation			
	Pinus radiata		*Betula alba*	
	2500 trees ha^{-1}	833 trees ha^{-1}	2500 trees ha^{-1}	833 trees ha^{-1}
Tree	41.58a	10.04bc	26.29ab	5.04c
Pasture + litterfall	7.92c	12.65ab	10.95bc	16.30a
Soil	50.50b	77.31a	62.75ab	78.66a

Please cite this article in press as: Fernández-Núñez, E., et al., Carbon allocation dynamics one decade after afforestation with *Pinus radiata* D. Don and *Betula alba* L. under two stand densities in NW Spain. Ecol. Eng. (2010), doi:10.1016/j.ecoleng.2010.03.007

(better climatic conditions and soil fertility) and age of the stands analysed in New Zealand (Lavery, 1986). In regard to the birch, the carbon sequestration capacity we obtained is similar to that found in the Nordic countries of Europe (Karlsson et al., 1998). In our case, a decrease in forest productivity occurs due to summer drought, whereas the Nordic countries experience a similar decrease in productivity during the winter cold.

The sequestration of C when a tree component is present is also impacted by the density of the established plantation. In this study, the difference in C sequestration capacity increased by 28% and 18% in pine and birch, respectively, at the higher density. When establishing a silvopastoral or forest system in an agricultural zone with no competition between trees, there is a direct relationship between the system's capacity for C sequestration and the plantation density. However, in the future, this capacity could be limited as the competition among trees increases and limits growth.

During the first years, pasture production in the systems was similar for both plantation densities, resulting in no effect of tree coverage on production. As a result of tree growth and, consequently, tree canopy, the microclimate conditions of the systems change and influence pasture production. Differences in plantation densities and among distinct ecological patterns of pine and birch are also known to influence pasture production (Sibbald et al., 1991; Silva-Pando et al., 2002). Likewise, as the system develops, the C sequestration capacity of the pasture diminishes, as the biomass production is reduced when lower amounts of light are able to reach the herbaceous layer. In our case, the amounts of C sequestration recorded aboveground on pastures show that silvopastoral systems in which the parameters of tree cover (growth rate, crown shape, deciduous leaves, needles) allow the pasture to expand and maintain a high production rate over time have a higher accumulation rate of carbon in this component. When comparing pasture production under pine versus under birch, we find that production under birch is higher due to the more light that reaches the understory, the slower growth rate of the tree, and the canopy shape. Therefore, these factors contribute to the higher capacity for carbon absorbed by the pasture component under birch cover.

The reduction of C sequestration capacity in the pasture can be compensated for by the C accumulation in tree litter on the forest floor in the systems with pines planted at high densities (Vesterdal et al., 2002). This result is consistent with our findings in those systems established under higher density stands (2500 trees ha^{-1}), where the decrease in pasture production that occurred under pine cover, as well as the consequently lower C content that accumulated, was partially compensated for by the carbon accumulation in the litter layer of these systems. Ultimately, there was a 25% reduction in C accumulation when compared to the levels accumulated in pastures growing under birch (1.9 and 2.5 Mg C ha^{-1} $year^{-1}$ under pine and birch cover, respectively). The results were nearly identical in the lower tree density systems (833 trees ha^{-1}) in that the accumulation of carbon in the litter under pine (2.03 Mg C ha^{-1} $year^{-1}$) was also 25% lower than that in the pasture component under birch (2.71 Mg C ha^{-1} $year^{-1}$). The presence of herbaceous pasture or litterfall in our system will have varying effects on the rate of the carbon incorporation of these residues into the soil. In forest systems, litterfall on the soil surface is the primary organic input, but in many cropping and grassland systems, the primary organic input is the decomposition of the roots and senescent pasture material (Gale and Cambardella, 2000).

On the other hand, the higher pasture production occurring in the birch systems could, in turn, provoke a larger production of GHG from the livestock, and thereby, a greater pasture carrying capacity than could otherwise be sustained. The C emitted by the animals translates, in all of the treatments, into 40% of the carbon stored in the herbaceous component. The reduced pasture carrying capacity that could be sustained by the silvopastoral systems considered in this study, in comparison with exclusively pastoral systems within this zone (Mosquera and González, 1998), result in less estimated emissions that are also compensated by the sequestration of C in the other components of the system (tree and soil). This implies that the C emissions on the part of the ruminants can be related to the use of animal stocking rates that are neither adjusted to the production capacity of the system, nor related to the elevated pasture carrying capacity and animal production that occurs in systems that are not based on pasture production. In other words, it is based on stabling of the animals or intensive farming.

Soil is the final destination for the majority of carbon fixed by photosynthesis in the Earth's ecosystems, and can be a major sink of atmospheric CO_2 (Lal, 2004). Furthermore, this soil carbon, in many forest systems, can remain stored for hundreds of years (Bouwman, 1990). Forest management, including a change in tree species and density, has been accepted as a measure of mitigation of atmospheric CO_2 in national greenhouse gas budgets (Vesterdal et al., 2008). However, quantitative estimates of tree species effects on soil C pools are still scarce (Vesterdal et al., 2008). Soil carbon sequestration in a silvopastoral system depends, among other things, on (i) organic matter inputs from pasture and tree residues, (ii) the litterfall quality and quantity from the tree and pasture, and (iii) the mineralisation rate, which depends on soil chemical characteristics, like the pH, and environmental factors, like temperature and humidity, which are also affected by tree species. In our study, establishing a forest on abandoned agricultural land with a nearly neutral pH caused an increase in acidity in the soil 11 years after planting (Mosquera-Losada et al., 2006). Low soil pH may inhibit litter decomposition and the incorporation of litter C into soil organic carbon (Sauer et al., 2007). Thus, in our case, few differences were detected between pine and birch in relation to soil carbon accumulation after 11 years, but, both have higher final C storage than in the initial conditions. However, SOC sequestration in deeper soil layers could be more important under *P. radiata* than *B. alba* due to better coarse root development (Fontaine et al., 2007; Li et al., 2007). Studies carried out in the same experiment in 2007 (Howlett, 2009) revealed that around 25% of organic carbon were placed between 25 and 1 m of depth, which means that most of the SOC was in the first 25 cm as found Jiménez et al. (2008) in dry tropical forests. No significant differences on total SOM concentration between densities or tree species were found in the 25–50, 50–75 and 75–100 soil depth layers (Howlett, 2009). Moreover, the proportion of fine roots, main source of SOM in deeper soil layers were also very low in this experiment (Howlett, 2009).

Vesterdal et al. (2002) and later Guo et al. (2007) have noted that the decrease in C in the surface soil layer after afforestation was partially offset by C accumulation in tree litter on the forest floor. For us, this compensation was more marked for the conifer than the deciduous trees used in the study due to the higher growth rate of the pine compared to the birch (Mosquera-Losada et al., 2006; Moreno and Pulido, 2009). Differences on accumulation in tree litter on the forest floor tended to disappear with age, among the pine stand of different densities. Furthermore, with conifers at the densities used, there was higher acidification, resulting from the closing of the canopy that led to subsequent needle death and loss, particularly in the lower branches. This incorporation of acidic substances into the edaphic material (Mosquera-Losada et al., 2006) reduces the rate of litterfall incorporation in the soil under this species.

The initial C content in the soil of our study was within the range of grassland soils (Calvo de Anta et al., 1992), while at the end it was close to those established (Macías et al., 2001) for Umbrisol forest soils in Galicia (125–187.5 Mg C ha^{-1}). Planting trees on soils pre-

Please cite this article in press as: Fernández-Núñez, E., et al., Carbon allocation dynamics one decade after afforestation with *Pinus radiata* D. Don and *Betula alba* L. under two stand densities in NW Spain. Ecol. Eng. (2010), doi:10.1016/j.ecoleng.2010.03.007

viously managed for crop or forage production has the potential to significantly alter soil properties (Paul et al., 2002). The distinct rates of organic matter production that depend on the tree type and density found in our case eventually influence soil organic carbon (Lugo and Brown, 1993; Guo and Gifford, 2002). Each species (broadleaf and conifer) has a different carbon allocation strategy that results in a different pattern, rate, quality, and quantity of organic carbon input to the soil (Lugo and Brown, 1993; Guo and Gifford, 2002). In our case, systems established under birch tended to demonstrate a greater rate of C accumulation and storage in the soil compared to those established under pine at the higher densities, despite the notably inferior rate of forest production (Lal et al., 1995).

Large differences were found in the annual system balance of carbon sequestration in the studied systems being more important for pines. There were also appreciable differences in the allocation of carbon to the different components of the systems studied. In any case, systems under densely planted conifers had a major proportion of carbon in the tree, compared to broadleaf stand and to lower density pine stand. Since differences in the global balance of carbon were found, it is clear that carbon stored in this system would remain shorter time in this area because, once the timber is harvested, potentially 50% of the system's carbon could be extracted from this type of forestland. This would not occur in the case of low-density treatments in which the majority of carbon is stored in the soil and is, consequently, more enduring. It is important to note that this differential division of carbon will cause differences in forest management decisions regarding carbon balance. After a thinning, the reduction of stored carbon in a high-density plantation of conifer would be directly affected by the removal of those trees, and in the case of the low-density plantation, by the effect that the removal of trees would have on the soil. The highest production of pasture occurred under deciduous trees at low density, which also had the highest accumulation of carbon in the soil due to the fast integration of the leaf into the soil. This difference was compensated for, however, by a higher accumulation of carbon in the tree in the case of the higher density pine plantations as was described by Palma et al. (2006) which indicates that the main difference in sequestration between an arable system and an agroforestry system lies in the carbon immobilized in the tree biomass.

In our region, agroforestry systems planted under deciduous trees at low-density result in the highest compatibility with animal production, since the deciduous trees allow for higher pasture production and, therefore, a higher annual profitability for the landowner. Even though, global levels of carbon sequestration in birch were lower than in pines, the storage of C was more linked to the soil in the deciduous low-density tree plantations, which results in a more enduring storage capacity. This has a notable socio-economic impact if environmental, as opposed to low quality wood production issues, are taken into account for afforestation policies.

In conclusion, at the end of 11 years, the establishment of an agroforestry system resulted in an increase in carbon sequestration capacity. We found that tree density first and forest species secondly had significant impacts on the differential capacity to sequester carbon within the system. The largest stock of carbon was found in the trees in all cases, with the exception of the birch systems at the lower density. This resulted in a significant difference in the amount of GHG emissions by the livestock if the pasture carrying capacity was adjusted to pasture production, or in other words, with extensive systems.

On the other hand, reforestation with low-density birch rather than pine would generate higher edaphic C sequestration rates, while still allowing for reasonable pasture production.

Uncited reference Q3

Stephan et al. (2000).

Acknowledgements

This paper has been completed thanks to the financial assistance of the Spanish Ministry and Xunta de Galicia. The authors would like to thank Teresa López Piñeiro, José Javier Santiago Freijanes, Divina Vázquez Varela, Pablo Fernández Paradela and Mónica García Fernández for their collaboration in the realisation of this study.

References

Bouwman, A.F., 1990. Exchange of greenhouse gases between terrestrial ecosystems and the atmosphere. In: Bouwman, A.F. (Ed.), Soils and the Green House Effect. Wiley, Chischester, pp. 61–127.

Bunker, D.E., DeClerk, F., Bradford, J.C., Colwell, R.K., Perfecto, I., Phillips, O.L., Sankaran, M., Naeem, S., 2005. Species loss and above-ground carbon storage in a tropical forest. Science 310 (5750), 1029–1031.

Calvo de Anta, R., Macías, F., Riveiro Cruz, A., 1992. Aptitud agronómica de la provincia de La Coruña (cultivos, pino, roble, eucalipto y castaño). University of Santiago de Compostela, Spain (in Spanish).

Chang, S.X., Mead, D.J., 2003. Growth of radiate pine (*Pinus radiata* D. Don) as influenced by understory species in a silvopastoral system in New Zealand. Agroforest. Syst. 59, 43–51.

Dixon, R.K., 1995. Agroforestry systems: sources or sinks of greenhouse gases? Agroforest. Syst. 31, 99–116.

EC, 2005. Communication on the implementation of the EU Forestry Strategy. In: Commission Staff Working Document (accessed 1.06.09.) http://ec.europa.eu/agriculture/publi/reports/forestry/workdoc_en.pdf.

EEA (European Environment Agency), 2003. Europe's environment: the Third Assessment. EEA, Copenhagen, http://reports.eea.europa.eu/environmental_assessment_report_2003_10/en/kiev_chapt_00.pdf.

EU, 1992. COUNCIL REGULATION (EEC) No 2080/92 of 30 June 1992 Instituting a Community aid Scheme for Forestry Measures in Agriculture (accessed 15.05.09.) http://www.legaltext.ee/text/en/T30207.htm.

EU, 2005. Council regulation (EC) n° 1698/2005 of Septembre 2005 on Support for Rural Development by the European Agricultural Fund for Rural Development (EAFRD) (accessed 15.05.09.) http://eurlex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2005:277:0001:0040:EN:PDF.

FAO-ISRIC-ISSS, 1998. World Referente Base for Soil Resources. World Soil Resources Reports 84. FAO, Rome.

Fernández-Núñez, E., Rigueiro-Rodríguez, A., Mosquera-Losada, M.R., 2007. Economic valuation of different land use alternatives: forest, grassland and silvopastoral systems. Grass. Sci. Eur. 12, 508–511.

Fontaine, F., Barot, S., Barré, P., Bdioui, N., Mary, B., Rumpel, C., 2007. Stability of organic carbon in deep soil layers controlled by fresh carbon supply. Nature 450, 277–280.

Flores Clavete, G., Gónzalez Arráez, A., Díaz Núñez, M., 1992. Producción ovina sobre praderas de zona costera de Galicia: efecto del sistema de pastoreo (rotacional y continuo) y de tres niveles de intensidad de pastoreo sobre la producción de pasto y producción animal. Xunta de Galicia, Spain (in Spanish).

Gale, W.J., Cambardella, C.A., 2000. Carbon dynamics of surface residue- and root-derived organic matter under simulated no-till. Soil Sci. Soc. Am. J. 64, 190–195.

Gómez-Rey, M.X., Calvo de Anta, R., 2002. Datos para el desarrollo de una red integrada de seguimiento de la calidad de suelos en Galicia (N.O. de España): Balances geoquímicas en suelos forestales (*Pinus radiata*). 1. Aportes de elementos por deposición atmosférica y hojarasca. Edafología 9 (2), 81–196.

Gordon, A.M., Naresh, R.P.F., Thevathasan, V., 2005. How much carbon can be stored in Canadian agroecosystems using a silvopastoral approach? In: Mosquera-Losada, M.R., McAdam, J., Rigueiro-Rodríguez, A. (Eds.), Silvopastoralism and Land Sustainable Management. CAB International, Wallinford, pp. 210–219.

Guitián-Ojea, F., Carballás-Fernández, T., 1976. Técnicas de análisis de suelos. Pico Sacro, Spain (in Spanish).

Guo, L.B., Cowie, A.L., Montagu, K.D., Gifford, R.M., 2007. Carbon and nitrogen stocks in a native pasture and an adjacent 16-year-old *Pinus radiata* D. Don plantation in Australia. Agric. Ecosyst. Environ. 124 (3–4), 205–218.

Guo, L.B., Gifford, R.M., 2002. Soil carbon stocks and land use change: a meta analysis. Glob. Change Biol. 8 (2), 345–360.

Howlett, D., 2009. Environmental amelioration potential of silvopastoral agroforestry systems of Spain: soil carbon sequestration and phosphorus retention. PhD Thesis. University of Florida, USA, p. 178.

IPCC (IntergovernmentalPanel on Climate Change), 1996. Reporting Instructions Guidelines for National Greenhouse Gas Inventory, vol. 2. Intergovernmental Panel on Climate Change, http://www.ipcc.ch/about/index.htm.

Jiménez, J.J., Lal, R., Leblanc, H.A., Russo, R.O., Raut, Y., 2008. The soil C pool in different agroecosystems derived from the dry tropical forest of Guanacaste, Costa Rica. Ecol. Eng. 34, 289–299.

Karlsson, A., Albrektson, A., Forsgren, A., Svensson, L., 1998. An analysis of successful natural regeneration of downy and silver birch on abandoned farmland in Sweden. Silva Fenn. 32 (3), 229–240.

Kirby, K.R., Potvin, C., 2007. Variation in carbon storage among tree species: implications for the management of a small-scale carbon sink project. Forest Ecol. Manage. 246 (2–3), 208–221.

Lal, R., 2004. Soil carbon sequestration impacts on global climate change and food security. Science 304, 1623–1627.

Lal, R., Kimble, J.M., Levine, E., Stewart, B.A., 1995. World soils and greenhouse effect: an overview. In: Lal, R., Kimble, J.M., Levine, E., Stewart, B.A. (Eds.), Soils and Global Change. CRC Lewis Publishers, Boca Raton, pp. 1–7.

Lavery, P.B., 1986. Plantation forestry with *Pinus radiata*. In: Review Papers No 12. School of Forestry, University of Canterbury, New Zealand.

Li, Z.P., Han, F.X., Su, Y., Zhang, T.L., Sun, B., Monts, D.L., Plodinec, M.J., 2007. Assessment of soil organic and carbonate carbon storage in China. Geoderma 138, 119–126.

Lugo, A.E., Brown, S., 1993. Management of tropical soils as sinks or sources of atmospheric carbon. Plant Soil 149, 27–41.

Maia, S.M.F., Alisson, S.X.F., Senna Teógenes, O., Eduardo, S.M., Filho, J.A., 2007. Organic carbon pools in a Luvisol under agroforestry and conventional farming systems in the semi-arid region of Ceará. Braz. Agroforest. Syst. 71 (2), 127–138.

Macías, F., Calvo, R., Arce, F., Bulnes, C., López, R., 2001. Los suelos como sumidero de carbono: materia orgánica de los suelos de Galicia. XXII Reunión Nacional de la Sociedad Española de la Ciencia del Suelo. XXII Reun. Nac. SECS. Dpto. Edafología, pp. 118–121.

Merino, A., Rey, C., Brañas, J., Rodríguez-Soalleiro, R., 2003. Biomasa arbórea y acumulación de nutrientes en plantaciones de *Pinus radiata* D. Don en Galicia. Invest. Agrar.: Sist. Recur. For. 12 (2), 85–98.

Montero, G., Ruiz-Peinado, R., Muñoz, M., 2005. Producción de biomasa y fijación de CO2 por los bosques españoles. INIA, Serie forestal, n° 13, Madrid, Spain, (in Spanish).

Moreno, G., Pulido, F.J., 2009. The function, management and persistence of Dehesas. In: Rigueiro-Rodríguez, A., McAdam, J., Mosquera-Losada, M.R. (Eds.), Agroforestry in Europe: Current Status and Future Prospects. Advances in Agroforestry. Springer, Berlin, pp. 127–160.

Mosier, A.R., Kroeze, C., Nevison, C., Oenema, O., Seitzinger, S., Van Cleemput, O., 1998. Closing the global N_2O budget: nitrous oxide emissions through the agricultural nitrogen cycle. Nutr. Cycl. Agroecosyst. 52 (2–3), 225–248.

Mosquera, M.R., González, A., 1998. Effect of annual stocking rates in grass and maize + rye systems on production by dairy cows. Grass Forage Sci. 53, 95–108.

Mosquera-Losada, M.R., Fernández-Núñez, E., Rigueiro-Rodríguez, A., 2006. Pasture, tree and soil evolution in silvopastoral systems of Atlantic Europe. Forest Ecol. Manage. 232, 135–145.

Mosquera-Losada, M.R., McAdam, J., Romero-Franco, R., Santiago-Freijanes, J.J., Rigueiro-Rodríguez, A., 2009. Definitions and components of agroforestry practices in Europe. In: Rigueiro-Rodríguez, A., McAdam, J., Mosquera-Losada, M.R. (Eds.), Agroforestry in Europe Current Status and Future Prospects. Advances in Agroforestry. Springer, Berlin, pp. 3–20.

Nair, P.K.R., Gordon, A.M., Mosquera-Losada, M.R., 2008. Agroforestry. In: Jorgensen, S.E., Fath, B.D. (Eds.), Ecological Engineering, Encyclopedia of Ecology. Elsevier, Oxford, pp. 101–110.

Nair, V.D., Haile, S.G., Michel, G.A., Nair, P.K.R., 2007. Environmental quality improvement of agricultural lands through silvopasture in southeastern United States. Sci. Agric. 64 (5), 513–519.

Palma, J.H.N., Graves, A.R., Burgess, P.J., Deesman, K.J., van Keulen, H., Mayus, M., Reinser, Y., Herzog, F., 2006. Methodological approach for the assessment of environmental effects of agroforestry at the landscape scale. Ecol. Eng. 29 (4), 450–462.

Paul, K.I., Polglase, P.J., Nyakuengama, J.G., Khanna, P.K., 2002. Change in soil carbon following afforestation. Forest Ecol. Manage. 168, 241–257.

Porta-Casanellas, J., Roquero De Laburo, C., López-Acevedo, M., 2003. Edafología para la agricultura y el medio ambiente. Mundi Prensa, Madrid, Spain (in Spanish).

Prescott, C.E., Zabek, L.M., Staley, C.L., Kabzems, R., 2000. Decomposition of broadleaf and needle litter in forests of British Columbia: influences of litter type, forest type, and litter mixtures. Can. J. For. Res. 30 (11), 1742–1750.

Reynolds, P.E., Simpson, J.A., Thevathasan, N.V., Gordon, A.M., 2007. Effects of tree competition on corn and soybean photosynthesis, growth, and yield in a temperate tree-based agroforestry intercropping system in southern Ontario, Canada. Ecol. Eng. 29, 362–371.

Rigueiro-Rodríguez, A., Mosquera-Losada, M.R., Gatica-Trabanini, E., 2000. Pasture production and tree growth in a young pine plantation fertilization with inorganic fertilisers and milk sewage in northwestern Spain. Agroforest. Syst. 48, 245–256.

Sauer, T.J., Cambardella, C.A., Brandle, J.R., 2007. Soil carbon and tree litter dynamics in a red cedar–scotch pine shelterbelt. Agroforest. Syst. 71, 163–174.

SAS, 2001. SAS/Stat User's Guide: Statistics Ed. SAS Institute Inc, Cary NC, USA.

Sibbald, A.R., Griffiths, J.H., Elston, D.A., 1991. The effects of the presence of widely spaced conifers on under-storey herbage production in the UK. Forest Ecol. Manage. 45 (1–4), 71–77.

Silva-Pando, F.J., González-Hernández, M.P., Rozados-Lorenzo, M.J., 2002. Pasture production in a silvopastoral system in relation with microclimate variables in the Atlantic coast of Spain. Agroforest. Syst. 56, 203–211.

Steinfeld, H., Gerber, P., Wassenaar, T., Castel, V., Rosales, M., De Haan, C., 2006. Livestock's Long Shadow, Environmental Issues and Options. LEAD-FAO, Roma.

Stephan, A., Meyer, A., Schmid, B., 2000. Plant diversity affects cultural soil bacteria in experimental grassland communities. J. Ecol. 88, 988–998.

Vesterdal, L., Ritter, E., Gundersen, P., 2002. Change in soil organic carbon following afforestation of former arable land. Forest Ecol. Manage. 169, 137–147.

Vesterdal, L., Schmidta, I.K., Callesen, I., Nilssona, L.O., Gundersen, P., 2008. Carbon and nitrogen in forest floor and mineral soil under six common European tree species. Forest Ecol. Manage. 255 (1), 35–48.

Xunta de Galicia, 2004. Inventario de emisións de gases de efecto invernadoiro en Galicia. Consellería de Medio Ambiente, Xunta de Galicia, Spain (in Spanish).

Zea-Salgueiro, J., 1992. Producción de ovino de carne en Galicia. Memorias del Centro de Investigación, Xunta de Galicia, Spain (in Spanish).