

1 Influence of tree species on carbon sequestration in 2 afforested pastures in a humid temperate region

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14 Abstract

15 Objectives. This study examines the influence of tree species in relation to biomass and soil C dynamics in
16 plantations established on former pasture land. Data on the C sink capacity of such plantations will provide
17 valuable information for designing improved management strategies for afforestation programmes aimed at
18 mitigating CO₂ emissions.

19 Methods. The study was carried in the temperate forest of southern Europe, one of the most productive timber
20 production systems in Europe. The study, designed to control most of the variability at regional level, involved a
21 network of 120 paired plots (former pasture land-new plantations of different ages) established to construct three
22 well-replicated chronosequences of the most common tree species in humid temperate systems.

23 Results. The mean rates of C sequestration (biomass and soil) estimated throughout the rotation ranged between
24 8.7 and 14.6 Mg C ha⁻¹ yr⁻¹ (*Eucalyptus nitens* > *Eucalyptus globulus* > *Pinus radiata*), and the contribution of
25 the soil (forest floor plus mineral soil) ranged from 8 to 18 % (*Eucalyptus nitens* > *Pinus radiata* > *Eucalyptus*
26 *globulus*). The humid temperate climate and the sandy loam texture of the soils favoured large losses of SOC
27 from the uppermost mineral soils during the 10 yr after afforestation. The higher loss of SOC in the *Pinus*
28 *radiata* soil (26% of initial SOC) than in the *Eucalyptus* soil (19.45% of initial SOC) was attributed to the lower
29 transfer of organic C to the mineral soil, as a result of the lower litter decomposition rate and the lower
30 belowground litter input from associated vegetation. The rapid development of tree biomass favoured the
31 subsequent C sequestration in biomass and soils.

32 Conclusion. The C sink capacity of forest plantations can be maximized by elongating the rotation length and
33 adopting suitable management strategies for each species. This is especially important in intensive forest
34 plantations in which the high intensity of harvesting may prevent accumulation of SOC in the long term.

1 *Keywords: Eucalyptus, Pinus radiata, pasture, afforestation, tree biomass, forest floor, soil*
2 *organic matter*

3 **Introduction**

4 The change in land use that is taking place in many areas of the world has a major
5 impact on the world's terrestrial carbon (C) balance and is therefore a key factor in the
6 mitigation of climate change (IPPC 2007). In the tropics, extensive areas of primary forest are
7 being converted to agriculture land to meet the current food demand. However, in Europe,
8 North America and Russia, traditional agriculture practices have been abandoned and large
9 areas of cropland and pasture land are being transformed to forests. In Europe, this process
10 has been favoured by the grant scheme for promoting reconversion of agricultural land to
11 forest (EU 1992), with the aim of reducing surplus agriculture production (Stanturf and
12 Madsen 2002), although at the same time it may have contributed to enhancing the C sink
13 capacity of terrestrial ecosystems.

14 The rate of accumulation or loss of soil organic carbon (SOC) following land use
15 change is governed by the balance between C inputs and C decomposition rates. This balance
16 becomes negative as the intensity of land management increases from forest to grassland to
17 cropland. However, the magnitude and progress of the changes in C contents following
18 changes in land use are highly variable because of the influence of different factors, such as
19 net primary production and rooting patterns (Jobbágy and Jackson 2000), quality of organic
20 inputs (Berg 2000), certain soil properties (nutrient availability, texture (Golchin et al. 1994;
21 Mendham et al. 2003)) and the intensity of past and present management (Balesdent et al.
22 2000).

23 Although the conversion of natural forest land to cropland produces clear and rapid
24 losses of biomass C and SOC (Bashkin and Binkley 1998; Paustian et al. 2000; Post and
25 Kwon 2000; Schimel 1995; Schlesinger 1986; Smith et al. 1993; Vitorello et al. 1989; Walker
26 and Desanker 2004), the effect of the opposite change in land use on SOC is slower, and C
27 recovery is not universally observed, at least in the short term. Nevertheless, data on the
28 changes in SOC following conversion of forests to pasture are widely variable, and while
29 some authors observed reductions in SOC of 20-30% (Detwiler 1986; Glaser et al. 2000;
30 Guggenberger and Zech 1999; Rhoades et al. 2000; Veldkamp 1994), others did not find
31 significant changes (Bell et al. 1995; Corre et al. 1999; Mendham et al. 2003; Reiners et al.
32 1994) and yet others reported increases (Fearnside and Barbosa 1998; Lugo and Brown 1993;

1 Neill et al. 1997). Murty et al (2002) and Guo & Gifford (2002) carried out a meta-analysis
2 worldwide, and reported that conversion of forest to uncultivated grazed pasture land does not
3 generally lead to loss of SOC, although losses or gains may be derived from the application of
4 fertiliser or removal of plant residues. This demonstrates the great potential of pastures to
5 sequester SOC, mainly because of the large inputs of C via herbaceous root litter (Guo et al.
6 2007), which may lead to more C accumulation in upper mineral soil than in forests (Jackson
7 et al. 2002).

8 There is great uncertainty about the changes in SOC associated with afforestation of
9 pasture land, because of the high SOC content of the latter. Afforestation of this type of land
10 usually leads to decreases in SOC in the short or medium term (Laganière et al. 2010; Poeplau
11 et al. 2011), and sometimes no variations are observed in the long term (Marín-Spiotta et al.
12 2009; Poeplau et al. 2011). This becomes more important when the species used in
13 afforestation are fast growing species with a short rotation length, in which recovery of initial
14 SOC losses may not occur by the end of the rotation. Some authors have suggested that this is
15 a temporary effect, in which only the labile C pool is exhausted (Cerli et al. 2008; Huang et
16 al. 2011a). However, it is not clear whether these initial losses are compensated in pastures
17 afforested with fast growing species.

18 The C balance after afforestation is greatly affected by the tree species, as a result of
19 differences in growth rates of the trees. In addition, litter production and litter quality, which
20 are greatly influenced by tree species, have a strong influence on the SOC dynamics (Berg
21 2000; Vesterdal et al. 2008). However, since most of the current knowledge is based on
22 studies of afforestation with coniferous species (Berthrong et al. 2009), the influence of the
23 tree species on SOC dynamics has not yet been accurately evaluated.

24 In studies that have attempted to evaluate the effect of tree species on afforested land,
25 the differences in SOC dynamics are attributed to the influence of the different turnover rates
26 of the litter (Vesterdal et al. 2008), the cover and type of ground vegetation (Lemma et al.
27 2006), or both (Huang et al. 2011a; Kasel et al. 2011; Paul et al. 2002). The influence of N-
28 fixing tree species has also been recognized (Kasel et al. 2011; Nilsson and Schopfhauser
29 1995), as a higher yield in poor soils leads to higher OM inputs to the soil, and also improves
30 litter quality and the speed of decomposition of OM (Conteh et al. 1997). Recent studies have
31 shown that these different sources of litter inputs can even lead to changes in the SOM
32 composition (Huang et al. 2011a).

33 The capacity of the soil to act as a C sink is also influenced by how rapidly the litter
34 converts C into humus (e.g. Kanerva and Smolander 2007; Prescott 2010; Silver et al. 2004).

1 The decomposability of the litter not only depends on its chemical composition, but also on
2 microclimatic conditions determined by the different canopy development and stand structure.
3 Moreover, the same microclimatic conditions have a direct influence on the cover and type of
4 ground vegetation, which in turn alters the amounts, composition and type of litter (Berg et al.
5 2009; Ostertag et al. 2008). This is particularly important because roots incorporate more C
6 into the soil than forest floor material (Jones et al. 2009). Thus, ground vegetation dominated
7 by grass species incorporates C rapidly into soil organic matter because the root system
8 develops quickly (Andrade et al. 2008; Laungani and Knops 2009).

9 The silvicultural parameters most closely related to SOC dynamics are: tree species,
10 site preparation techniques, initial stocking, rotation length and other parameters more
11 specific to each type of tree species management (pruning, clearcutting, thinning, application
12 of fertilizer, etc.) and autoecology. These aspects have been discussed by Jandl et al. (2007)
13 and, in the case of the species considered here, by Balboa-Murias et al. (2006). In addition to
14 management practices, a comprehensive representation of the entire forestry sector system
15 should be considered, taking into account the C pools and fossil fuel substitution, although the
16 latter pool has not been considered in the international agreements on reduction of emissions
17 (UNFCCC 2002). All of these factors are particularly important in fast growing tree species,
18 in which short rotations and harvesting of logging residues may prevent accumulation of SOC
19 in the long term (Huang et al. 2010b).

20 Most studies concerning the effect of different species and types of management on C
21 sequestration in afforested land have compared the C stocks in several pools in pasture land
22 and forest plantations in the steady state, rather than considering the dynamic changes that
23 take place. In many cases, the shifts in key parameters throughout the rotation, such as tree
24 growth, stand structure, associated vegetation and forest floor development are not
25 considered. Thus, the temporal dynamics not only provide an understanding of the different
26 mechanisms of C sequestration after afforestation, but are also useful for designing the most
27 appropriate type of management to maximize the C sink capacity.

28 Most approaches evaluating the capacity of soils as C sinks focus on plot level, and few
29 studies have been extended to landscape levels (Conant et al. 2003; Johnston et al. 1996;
30 Turner and Lambert 2000). Such studies show the high degree of variability in the SOC
31 dynamics following afforestation, even under rather homogeneous conditions. This high
32 variability emphasizes the risk of making erroneous conclusions about SOC dynamics when
33 the experimental design does not take this variability into account (Berg et al. 2009). A
34 correct methodology must ensure adequate sampling replication at plot level, and proper

1 sample analysis to take into account most of the variability for extrapolation of the results of
2 plot level studies to a larger scale (Goidts et al. 2009). Methodological procedures for
3 quantifying the changes in SOC after afforestation are: i) paired sites, ii) chronosequence
4 studies, iii) multiple re-sampling, and iv) process and modelling studies (Turner and Lambert
5 2000).

6 Chronosequence studies use a series of plots in plantations of different ages with
7 supposedly similar management regimes and environmental conditions, and translate spatial
8 differences between soils into temporal differences (Huggett 1998). Although the
9 chronosequence approach cannot replace long-term field experiments, there are certain
10 disadvantages with the latter, such as the delay in obtaining results, workload and particular
11 trends in external parameters (i.e. climate and local conditions), which may cause a systematic
12 bias in the observations. The paired-plots approach is an alternative method in which one of
13 the paired plots represents the initial conditions. Because of the high spatial variability in
14 SOC measurements (Conant et al. 2003; Johnston et al. 1996; Turner and Lambert 2000), the
15 combined use of chronosequences and the paired-plots approach may provide a suitable way
16 of detecting changes in soil pools, and of correcting local tendencies.

17 The objectives of the present study were: a) to examine the influence of tree species on
18 the C dynamics in the forest system (tree biomass, forest floor and mineral soil) following
19 afforestation of pasture land, b) to explore the relationships between tree biomass
20 development and forest floor and SOC dynamics in three forest plantations established on
21 former pasture land, and c) to evaluate the C sink capacity of the different types of plantations
22 in relation to management. A specific design combining chronosequence and paired plot
23 approach (afforested subplot and pasture subplot) was applied to provide accurate information
24 on C dynamics in the different tree species planted. The design was applied to temperate
25 forest plantations of southern Europe, one of the most productive timber production systems
26 in Europe, where important afforestation processes may play an important role in mitigating
27 CO₂ and other GHG emissions. The data obtained in this study will provide valuable
28 information about the effects of such afforestation programmes on C sink capacity.

29 **Materials and methods**

30 **Site description and experimental design**

31 The study was carried out in Galicia (NW Spain), in an area of 7000 km² representative
32 of the Atlantic-climate zone of northern Spain. The 20 year annual average rainfall in the area

1 is 1378 mm (range 884-2107 mm), and the temperature, 12.1°C (range 10.3-14.8°C). The
2 wettest month is November, with an average rainfall of 160 mm, and the driest August, with
3 44 mm. The lowest mean monthly temperature 7.1°C occurs in January, and the highest
4 18.2°C, in August. The soil humidity and temperature regimes are Udic (mean period with
5 partial drought, 1 month) and Mesic (mean frost-free period, 10 months), respectively. The
6 soils were developed from granitic rocks, schist and shale, and classified as Humic or Distric
7 Cambisols and Alumi-humic Umbrisols (IUSS Working Group WRB 2006). The soil has a
8 loam or sandy loam texture and is well drained.

9 The average values for selected characteristics of the afforested plantations studied are
10 shown in Table 1. The observed range of site qualities was higher than in plantations
11 established on former forest soils, probably because of the better quality of the soils (soil
12 depth, stoniness, higher water supply) and prior fertilization. A network of 120 paired plots,
13 made up of 120 pasture subplots adjacent to 120 afforested plantations in former pasture
14 plots, was established. The plots were distributed in three sets of 40 plots planted with the
15 most commonly used species in the area: *Eucalyptus globulus* Labill, *Eucalyptus nitens*
16 (Deane & Maiden) Maiden and *Pinus radiata* D. Don. Each set was an independent
17 chronosequence in which the range of ages covers the usual rotation lengths applied to these
18 plantations, thus enabling conclusions to be reached as regards the effects of the single-stem
19 rotation following land use change: 1-23 years for *E. globulus*, 2-18 years for *E. nitens* and 2-
20 40 years for *P. radiata* chronosequence respectively.

21 In all cases the prior use was as pasture land, in which low intensive management was
22 applied for at least 25 years (according to the landowners), and some of which has recently
23 been afforested. When the fertility levels were adequate, the plots were dominated by a
24 mixture of *Lolium multiflorum*, *Lolium perenne*, *Trifolium pratense*, *Trifolium repens* and
25 *Dactylis glomerata*, although as the time since last perturbation increased, *D. glomerata*,
26 *Agrostis capilaris* and *Holcus lanatus* became more predominant. The pastures are normally
27 harvested for silage 1-2 times a year, and grazed once or twice a year. They are usually
28 renewed every 8-10 years by rotovating to a depth of 20 cm. In all cases, afforestation was
29 carried out after ripping at 50 cm depth and planting along the row, so that soil disturbance
30 was considered low. No fertilization or weed control was carried out in the plantations. The
31 pastures were selected so that time since last tillage was more than four years. Two
32 dimensional restrictions were applied to the pairwise land uses selected: a minimum size of
33 each grassland and afforested plantation of 0.5 ha, and stands where a 50 x 50 m square plot

1 could be established 30 m inside the other land use border for both pasture and forest, to avoid
2 the edge effect.

3 To ensure that all sites were similar as regards soil type and land use, selection of the
4 study sites was based on direct observation of the terrain in adjacent pastures, and
5 consultations with local landowners. The plantation age was established using an increment
6 borer to sum the ring number for *P. radiata* and *E. nitens*, which was clearly visible and easy
7 to assign to yearly growth periods, and considering the planting date declared by the owner to
8 the Forestry Administration for *E. globulus*.

9 Similar forest management regimes, in terms of site preparation, source of seedlings,
10 and pruning and harvesting regimes, were applied in all afforested stands. Stands of *E.*
11 *globulus* were located in coastal areas at altitudes below 300 m.a.s.l., whereas *E. nitens* and *P.*
12 *radiata* stands were located in the inner area, generally between 300 and 500 m.a.s.l. The only
13 theoretical difference assumed among plots, within a given tree species, was therefore age
14 since afforestation.

15 **C determination in aboveground tree biomass**

16 For determining C density (Mg C ha^{-1}) in aboveground biomass, diameter at breast
17 height (to the nearest cm) and total height (to the nearest dm) were measured in all trees in
18 circular plots of radius 10 m. Dry weight of aboveground biomass was estimated using the
19 equations proposed by Merino et al. (2005) for *E. globulus*, by Pérez-Cruzado and Rodríguez-
20 Soalleiro (2011) for *E. nitens*, and by Balboa-Murias et al. (2006) for *P. radiata*
21 chronosequences. The C concentrations in each biomass fraction were those reported by the
22 latter authors, except for *E. nitens*, for which the values reported by Pérez-Cruzado et al.
23 (2011) were used.

24 **Soil sampling and analysis**

25 For sampling the soil (forest floor and mineral soil to 30 cm depth), a 20 m diameter
26 plot was established within each stand, at a distance of more than 30 m from the edge of the
27 stand. Five samples per plot were taken from between tree rows to minimize any disturbance
28 from the site preparation. The forest subplots were the same as established for C
29 determination in aboveground biomass.

30 For sampling the forest floor, 0.3 x 0.3 m squared frames were thrown at random within
31 each plot, on 5 occasions. Samples of forest floor were collected avoiding removal of the
32 mineral soil, and dried at 40°C until constant weight. The C contents in the forest floor were

1 measured for *E. nitens* stands (47.9 %C), whereas already published data were used for *E.*
2 *globulus* and *P. radiata* stands (48.7 and 46.9 %C respectively, Merino et al. (2005)).

3 For mineral soil, three soil layers were collected at depths of 0-5, 5-15 and 15-30 cm at
4 the same sampling points as for forest floor. The first two correspond to the A horizon, and
5 the deepest layer A2, AB or BA. Sub-samples of the mineral soil layer were collected with a 5
6 x 20 cm steel corer, and were combined to form one bulk sample per plot. These samples
7 were oven-dried at 40°C, sieved at 2 mm and the stoniness was determined. In the fine soil
8 fraction, total C and N were analyzed with a LECO Elemental analyzer, and soil particle
9 analysis in the upper 15 cm was performed by laser diffractometry, with a Mastersizer 2000
10 diffractometer.

11 At each sampling point, five soil core were collected with a 100 cm³ metal cylinder,
12 and were oven-dried at 105°C and weighed to determine the bulk density, following the
13 methodology of Blake and Hartge (1986). The C content in each layer was determined by
14 equation (1), where CD is the C density in each layer (Mg ha⁻¹), CC is the C concentration in
15 each layer (as a fraction of unity), Db is the bulk density (g cm⁻³), T is the thickness of each
16 layer (cm) and S is the stoniness (as a fraction of unity).

$$CD = CC \cdot Db \cdot T \cdot (1 - S) \cdot 100 \quad (1)$$

17 Evaluation of C sequestration in the forest system

18 The amounts of C in aboveground biomass and forest floor were considered as net gain
19 relative to pastures, and therefore only absolute values are reported or plotted against time
20 since afforestation. For modelling of aboveground biomass C changes with time since
21 afforestation, the Richards (1959) model was used to describe the relationship between
22 aboveground biomass C (C_w , Mg ha⁻¹) and stand age (t , yr), shown in equation (2). The model
23 for each species was fitted with the MODEL procedure of the SAS/ETS® system (SAS
24 Institute Inc 2004). The root of mean square errors ($RMSE$) and adjusted determination
25 coefficient ($Adjust. R^2$) were calculated for each model fit.

$$C_w = b_0 \cdot (1 - e^{-b_2 \cdot t})^{b_3} \quad (2)$$

26 Carbon sequestration in each mineral soil layer was evaluated in absolute terms as the
27 difference between the forest subplot CD_F and the pasture subplot CD_P (C absolute difference
28 CAD , Mg ha⁻¹), and in relative terms as the difference in percentage of C density (C relative
29 difference, CRD , % of initial C density), with equation (3). Both parameters were represented

1 in each plot against time since afforestation to evaluate the changes in soil C with time since
2 afforestation.

$$CRD = \frac{CD_F - CD_P}{CD_P} \cdot 100 \quad (3)$$

3 To describe changes in soil C over time, in previous studies on changes in mineral soil
4 C after secondary succession, an empirical modelling approach including the gamma function
5 was used successfully (Covington 1981; Zak et al. 1990). However, in the present study the
6 changes were expected to follow a different pattern, as we hypothesized that the C
7 equilibrium level is different in pasture than in afforested land, and the gamma is function
8 biologically inconsistent in such cases, because it provides an asymptotic value equal to the
9 intercept. Alternative models include more parameters than the gamma function, making
10 convergence of the parametric model fit difficult, although this will depend on the amount of
11 data available. In this case, parametric curve fitting procedures did not converge because of
12 the large number of parameters needed to fit a model that adequately captures the apparent
13 shape of the data. Thus, nonparametric fitting was carried out to describe the general trend in
14 the changes in forest floor and mineral soils. The LOESS procedure in the SAS/STAT (SAS
15 Institute Inc 2004) was used to divide residuals of the nonparametric curve into ten age
16 intervals, and the 95% confidence levels were determined.

17 Data were also analyzed by analysis of variance with the GLM procedure of SAS/STAT
18 (SAS Institute Inc 2004). The model applied was: $Y_{ij} = \mu_i + F_{ij} + \varepsilon_{ij}$, where Y_{ij} is the dependent
19 variable value for species i and factor j , μ_i is the mean for species i , F_{ij} is the effect of the
20 factor j and ε_{ij} is an error term. The dependent variables considered were CD , CRD , the initial
21 C density, site index and texture variables. As they were not normally distributed, the data
22 were transformed by normalized ranks before analysis. The factors considered were age
23 classes, soil type (pasture or afforestation) and the position of the plot as regards the CD
24 accumulation curve. The Tukey test was used to detect differences between means,
25 considered significant at $p < 0.05$.

26 **Results**

27 **Changes in the C accumulated in biomass throughout the rotation**

28 Results of non linear fit of aboveground biomass C density are shown in Table 2. All
29 parameters were significant at $p < 0.005$, and the models explained most of the variation (with

1 reference to $Adjust.R^2$ values). The average changes in the C accumulated in the tree
2 aboveground biomass for the three tree species studied throughout the rotation are shown in
3 Fig. 1. The tree growth rates followed the order $E. nitens > E. globulus \geq P. radiata$. The three
4 species continued accumulating C at high rates, even after the common rotation lengths in the
5 region (10-16 years for both *Eucalyptus* and 30-35 years for *P. radiata*), which indicates the
6 large potential of the biomass of these species as a C sink.

8 **Changes in C accumulated in the forest floor throughout the rotation**

9 The changes in the C accumulated in the forest floor (and mineral soil to depth 15 cm)
10 for each of the three species are shown in Fig. 2. The average trends throughout the rotation
11 were fitted by the nonparametric procedure described in Materials and methods. The three
12 species showed different patterns as regards the dynamics and the amounts of C accumulated
13 at the end of the rotation. Thus, in accordance with the higher growth rates of biomass, forest
14 floor accumulation occurred earlier (2 yr after establishment) in *E. nitens* than in *P. radiata*
15 and *E. globulus* (4-5 yr after forest establishment). The forest floor C accumulation rates
16 followed the order: $E. nitens > P. radiata > E. globulus$. Forest floor accumulation was lower
17 than expected in *E. globulus* stands, considering the high aboveground tree growth. The
18 correlations between the changes in crown biomass and forest floor layer dynamics were
19 different for each species, and were generally rather weak (n.s. for *E. globulus*; $R^2 = 0.40$ for
20 *E. nitens*, and $R^2 = 0.55$ for *P. radiata*).

21 Forest floor accumulation stabilized 10 years after afforestation by both *Eucalyptus*
22 species, whereas it continued to increase in mature *P. radiata* plantations. At the end of the
23 rotation, the average amounts of C in the forest floor ranged from 10.1 (*E. globulus*, 12 yr), to
24 24.8 Mg ha⁻¹ (*E. nitens*, 12 yr) and 50.9 (*P. radiata*, 35 yr).

25 **Changes in SOC in the mineral soils after afforestation**

26 The changes in SOC density relative to that of the paired pasture (CRD) in the mineral
27 soil for each soil depth layer, considering each species separately and together, are shown in
28 Table 3. In the first 10 years after afforestation, losses of C in the 0-15 cm layer were found in
29 all three tree species, ranging between -52.0% in *P. radiata* to about -0.2% in both species of
30 *Eucalyptus* in the first 5 years. However, the ANOVA only revealed significant changes in the
31 0-5 and 5-15 cm soil layers under *P. radiata*, which emphasizes the high variability in the
32 *Eucalyptus* stands. In the 11-15 yr period, the SOC contents were similar in afforested and

1 pasture soils for all species. Finally, in the stands older than 20 yr (only for *P. radiata*), the
2 SOC contents were significantly higher in the uppermost soil layer (0-5 cm) than in the other
3 layers. No such trend was detected in the 15-30 cm layer.

4 The average mineral soil C density in pastures and afforested stands, grouped in age
5 classes of 10 years and for the three mineral soil layers studied, are shown in Fig. 3. For direct
6 comparison of the C densities in each soil depth, the value of each soil layer was divided by
7 the corresponding depth ($\text{Mg ha}^{-1} \text{ cm}^{-1}$). The average SOC densities in pasture sub-plots were
8 constant, since there were no significant changes in C densities over time for any of the
9 species studied ($p < 0.001$). Changes in *CRD* and *CAD* were therefore only due to changes in
10 SOC densities in forest subplots, because C remained constant for a given soil layer in pasture
11 land. There were no significant differences in the distribution of SOC across soil depth either
12 between species or ages considered ($p < 0.001$). Nevertheless, there were significant
13 differences in *P. radiata* stands in the upper mineral soil layers, as previously reported.

14 The mean changes in *CAD* and the 95% confidence levels in the 0-15 cm mineral soil
15 (and in the forest floor) for each of the three species considered in this study throughout their
16 respective rotations are shown in Fig. 2. For calculation of the average trends and confidence
17 levels, the nonparametric procedure fitting described above was applied. This type of
18 representation enables consideration of the variability in the data. Significant SOC losses
19 were detected in the first 5-10 years after afforestation in *P. radiata* stands. The average
20 losses amounted to -10.1 Mg ha^{-1} (for 95% of confidence level, between -7.1 to -13.0), which
21 constituted an average loss of -24.1% of the initial SOC (for 95% of confidence level,
22 between $18.8 - 29.4 \%$). There were then large gains in soil C, coinciding with significant
23 accumulation of forest floor, reflecting a change in the environmental equilibrium between
24 decomposition and production. The average compensation age (the time at which the initial
25 SOC content is recovered) was 20 yr (for 95% of confidence level, the data ranged from 14 to
26 25 yr) and progressive gains occurred thereafter. Thus, at the end of the first rotation the mean
27 gain in SOC in these mineral soils was $+9.2 \text{ Mg ha}^{-1}$ (for 95% of confidence level, between
28 $+3.8$ and $+15.0 \text{ Mg ha}^{-1}$) relative to the pastures where they were established.

29 In the *Eucalyptus* stands, the mean changes in SOC were rather different from those of
30 *P. radiata*. Despite the high variability observed, a general trend was distinguished. The
31 initial SOC losses started later than in the *P. radiata* plantations (Fig. 2), and although they
32 were of lower intensity in terms of *CRD* (Table 2), the average *CAD* values were similar. In
33 the *E. nitens* stands, in which the forest floor accumulation started earlier, the SOC levels
34 remained rather close to the initial contents throughout the whole rotation, and losses were

1 slightly lower (-7.2 Mg ha⁻¹ (values ranged from +0.1 to -15.2 Mg ha⁻¹)). In *E. globulus*, a
2 slight recovery appeared to occur at the end of the rotation, but the maximum losses for this
3 species amounted to -9.8 Mg ha⁻¹ at age 7 years (+2.0 to -23.8 Mg ha⁻¹), leading to average
4 losses of -9.2 % (+0.7 % and -19.1 %) of the initial SOC.

5 Levels of C accumulation in the soil throughout the first rotation were calculated
6 considering the two soil pools jointly, forest floor and mineral soil to a depth of 30 cm (Fig.
7 4). Net gains in C were observed from the beginning of the rotation in the soils under *E.*
8 *nitens*, as a result of the large contribution of the forest floor in these stands. In contrast,
9 significant gains were observed in *E. globulus* and *P. radiata* 8-9 yr after afforestation (Fig.
10 4). In both *Eucalyptus* stands the soil C density stabilised 15 yr after afforestation, whereas in
11 the *P. radiata* stands the soil C accumulation was still increasing after the usual rotation
12 length (30 yr). Root biomass (including stump) was not considered in this study.

13 To evaluate the effects of site characteristics on SOC dynamics after land use change,
14 the plots were classified into two groups, according to whether the *CAD* was above or below
15 the average non-parametric level, as shown in Fig. 2. No differences in site index were
16 revealed by the unbalanced variance technique, although the greatest gains were
17 systematically observed in stands with the highest site index values (Table 4). Nonetheless,
18 there were significant differences in the initial C content between the upper and lower part of
19 the curve for plots of both *Eucalyptus* species, in which the greatest losses in the mineral soil
20 coincided with the highest initial C density, although this effect was not significant for *P.*
21 *radiata* (Table 4). Although soil texture is known to affect SOC dynamics, there were no
22 significant differences between the upper and lower part of the *CAD* curve for this parameter,
23 nor any apparent trend, probably because of the high homogeneity among plots.

24 The average C/N ratio in the pasture mineral soil (0-5 cm depth) was 12.4 (2.1, Std.
25 dev.) and, as expected, this did not change with respect to the age of the paired afforested
26 stands (data not shown). The C/N ratio in the *P. radiata* stands increased with age ($R^2 = 0.3$;
27 $p < 0.05$) reaching an average value of 18.2 at the end of the rotation, which differed
28 significantly from the values in the pasture soils ($p < 0.01$). In contrast, the C/N ratio in the
29 soils under *Eucalyptus* did not differ from those in the paired pastures.

30 **Changes in C in the forest system throughout the first rotation after** 31 **afforestation**

32 The average changes in C (Mg ha⁻¹ yr⁻¹) throughout the rotation, taking into account the
33 aboveground tree biomass, the forest floor and the mineral for each of the three species are

1 shown in Fig. 5. In order to analyse the effect of the rotation length on the C sink capacity, the
2 total amounts of C accumulated in aboveground tree biomass and the accumulation rates
3 considering three possible rotation lengths for each tree species are shown in Table 5.

4 The effectiveness of C sequestration followed the order: *E. nitens* > *E. globulus* ≥ *P.*
5 *radiata*. The components accumulated C according to the following order: above ground
6 biomass (94-96 %) > forest floor (3.4-10.8 %) > 0-15 cm upper soil mineral (0.17-0.96 %). At
7 the end of each rotation the C accumulated followed the order: *P. radiata* > *E. nitens* > *E.*
8 *globulus*.

9 In all three species the ecosystem C accumulation rates kept increasing, even in the
10 longest rotations (Table 5). The most important effect was found in the *Eucalyptus* stands in
11 which the C sink rate increased by 2.0-3.3 Mg C ha⁻¹ yr⁻¹. Prolongation of the rotation length
12 in these species also resulted in net C gains in the mineral soil.

13 **Discussion**

14 **Experimental design**

15 The present study involved 120 paired plots (40 per tree species) in a geographical area
16 of 7000 km², which is rather homogeneous from the point of view of climate and soil
17 properties for each species distribution considered. The experimental design, based on well-
18 replicated chronosequences combined with paired plots, enabled most of the variability and
19 local conditions to be controlled, and average regional trends in the C dynamics for the three
20 most common tree species used in afforestations in northern Spain to be determined. Despite
21 the rather homogeneous environmental conditions, the variability in the C dynamics after
22 afforestation was high, especially in the *Eucalyptus* chronosequences. Such high variability,
23 which has also been reported in other studies at landscape scale (Conant et al. 2003; Johnston
24 et al. 1996; Poeplau et al. 2011; Turner and Lambert 2000), emphasizes the risk of reaching
25 wrong conclusions about SOC dynamics when the experimental design does not take into
26 account most of the variability.

27 **Changes in C in aboveground biomass and forest floor**

28 The growth rates of the fast-growing tree species studied differed widely. The mean
29 annual increments were higher than the productivity recorded for the plantations established
30 on former forest soils in the region (Balboa-Murias et al. 2006). The enhanced growth is
31 attributed to the greater depth and availability of nutrients and water in the pasture soils.

1 At the end of the rotation, the forest floor constituted between 3 and 11% of the forest
2 system C pool. The C sequestration rates in the forest floor (0.9-2.9 Mg C ha⁻¹ yr⁻¹) were
3 considerably higher than those reported for Central Europe and Scandinavian Countries ((Berg
4 et al. 2009; Vesterdal et al. 2002) lower than 0.5 Mg C ha⁻¹ yr⁻¹) and lower than in tropical
5 forests (Ostertag et al. (2008), 5 Mg C ha⁻¹ yr⁻¹).

6 The different forest floor accumulation rates associated with the three species only
7 partly correspond to the tree growth rates. Thus, the high forest floor accumulation in *E.*
8 *nitens* stands was possibly associated with the particularly high growth rate of this species
9 (Fig. 1). However, forest floor accumulation was lower in the *E. globulus* stands than in *P.*
10 *radiata* stands, despite the greater increase in biomass. Moreover, within the same species, the
11 relationships between the changes in tree biomass and forest floor were weak, indicating that
12 forest floor accumulation was also determined by other factors, such as decomposition rates
13 or changes in ground cover.

14 Thus, the lower forest floor accumulation in *E. globulus* may be due to the faster
15 decomposition of *Eucalyptus* litter than of *P. radiata* litter, as recorded in the region (Alvarez
16 et al. 2008) and elsewhere (Huang et al. 2011a; Lemma et al. 2007; Paul and Polglase 2004),
17 and attributable to differences in the chemical composition of the litter. Litter decomposition
18 in *Eucalyptus* may also have been enhanced by favourable microclimatic conditions. Light
19 transmission is higher in *Eucalyptus* than in *P. radiata* stands, because of the arrangement of
20 the leaves (González-Hernández et al. 1998), which probably leads to higher soil temperatures
21 and more favourable moisture levels (Martius et al. 2004b). This may have promoted
22 decomposer and macrofauna activities and thus higher decomposition rates (Martius et al.
23 2004a).

24 Nevertheless, some climatic effect can be confounded with species effects regarding
25 forest floor C accumulation. With respect to species distribution, *E. globulus* stands are
26 located at lower altitudes (lower than 300 m), where temperatures are less extreme, than *E.*
27 *nitens* stands. Autumn litterfall peak occurs earlier in *E. globulus* than in *E. nitens* stands, and
28 since forest floor samples were collected at the same time for all species (December-January),
29 *E. nitens* forest floor C accumulation may have been overestimated. Moreover, the litter
30 decomposition rate may also be affected by climatic differences associated with the
31 distribution of *E. globulus* and *E. nitens*.

32 With the exception of *E. nitens* stands, in which forest floor accumulation occurred very
33 early on, in *E. globulus* and *P. radiata* stands, development of the forest floor coincided with

1 canopy closure (around 5 yr after establishment). This implied an increased input of litter, as
2 well a lower mineralization rate as a consequence of shading.

3 In addition to these factors, ground vegetation was rather abundant in these young
4 plantations (3-8 Mg ha⁻²; Omil et al. (2007) and Carneiro et al. (2009)), but decreased steadily
5 with the increased shading throughout stand development (Fernández-Núñez et al. 2010).
6 Thus, canopy closure may have resulted in a sharp decrease in ground vegetation root
7 turnover. This may have been greater under *P. radiata* because of the greater degree of shade
8 provided by pines than by *Eucalyptus* (despite the usual tree density in the latter plantations).
9 Thus, Omil et al. (2007) recorded a decrease of 50 % in the ground vegetation during the first
10 10 yr after the establishment of a *P. radiata* plantation. On the other hand, the death of ground
11 vegetation as a result of shading in *P. radiata* stands contributes to the forest floor depth.
12 Higher litter production in association with the ground vegetation has also been reported
13 elsewhere (Stendahl et al. 2010).

14 **Changes in C in mineral soil**

15 The plantations under study replaced pastures growing on soils with rather high SOC
16 contents (74.8 Mg ha⁻¹ to a depth of 30 cm, 27.7 Mg ha⁻¹ Std. dev.), intermediate between
17 croplands and natural forest soils in the region (Leirós et al. 2000; Merino et al. 2004). The
18 high SOC contents are due to long use under low intensive management based on low
19 intensity tillage and addition of slurry, lime and fertilizers, which enhance grass production
20 and therefore root turnover. Other researchers have indicated the potential of grasslands to
21 sequester SOC, in some cases at levels comparable to those in forests (Corre et al. 1999;
22 Jackson et al. 2002).

23 The data obtained in this study revealed important losses of SOC in the upper mineral
24 soils during the first 10 years after afforestation, and subsequent gains, which may lead to
25 recovery of the initial levels after 20 years. The maximum SOC losses in the first 10 years
26 after afforestation in the upper 0-15 cm mineral soil layer were dependent on the tree species
27 and constituted 4 - 38 % of the initial C contents. Such high losses are common in afforested
28 grassland soils under humid temperate climates (Kirschbaum et al. 2008) and occur as a
29 consequence of mineralization of the high content of the labile fraction in these soils, which
30 was not compensated by decreased litter inputs from grass litter (Don et al. 2009; Huang et al.
31 2011a). In some cases, they are caused by soil perturbation during site preparation work
32 (Maillard et al. 2010; Turner and Lambert 2000). Although the soil was slightly perturbed as a
33 result of site preparation in the present study (the proportion of disturbed soil was 13% with

1 respect to the total area of the plot), it must be considered that the C losses may have been
2 underestimated because subsoiling lines were not sampled.

3 No corrections were made for changes in bulk density, since SOC_s were estimated for a
4 constant volume rather than for constant weight. In the afforested sub-plots, the C density
5 may have been underestimated as a result of considering a constant depth rather than a
6 constant volume (Post and Kwon 2000; Mendham et al. 2003; Poeplau et al. 2011; Toriyama
7 et al. 2011), as soil density was lower in afforested sub-plots than in pasture sub-plots in the
8 three species considered, for the 0-5 cm soil layer (0.93 to 0.79, 0.85 to 0.79 and 1.0 to 0.84
9 kg for *E. globulus*, *E. nitens* and *P. radiata* respectively). Since no changes in bulk density
10 were detected in deeper soil layers, the degree of underestimation was not considered to be
11 relevant.

12 In addition to the favourable climatic conditions, the high SOC losses in these soils may
13 also be caused by certain soil properties that favour the rapid turnover of SOM. The soils
14 under study contained low percentages of clay (less than 20 %; the texture is sandy loam),
15 which is dominated by minerals of low surface reactivity (kaolinite and oxides of Al and Fe).
16 Therefore, the capacity of these soils to stabilize C in mineral-associated forms and in fine
17 pores may be limited. Nevertheless, chemical stabilization due the saturation of Al and Fe in
18 the cation exchange complex (Hobbie et al. 2007) should also be considered. Different studies
19 have revealed that SOM turnover is usually rapid in the uppermost soil horizons, but takes
20 longer as the soil depth increases (e.g. von Lützow et al. 2006). It is assumed that SOM
21 stabilization mechanisms (physical inaccessibility to microorganisms and enzymes or
22 interaction with mineral surfaces and metal ions) are less active in the uppermost layers of
23 mineral soil than in deeper soil horizons (e.g. von Lützow et al. 2006).

24 Similar patterns to those described above have been described in other studies (Table 6).
25 These studies were selected to represent the same land use change as described in the present
26 study, and to provide information enabling estimates in the short and long-term and at
27 compensation age. In the short term (<20 yr), the average losses were similar to those
28 observed in the present study. The average C compensation ages recorded in the present
29 study, between 10 and 25 years, were within the most common range reported in the relevant
30 literature (Table 6), although much longer compensation ages (80 yr) have been simulated in
31 colder climates than described here. Moreover, greater (although highly variable) long-term
32 gains than those observed here have been reported, although the time frame considered was
33 longer than in the present study (Table 6).

1 The present results show that the subsequent gains in SOC took place after canopy
2 closure in the stands. Significant C gains were only recorded in the 15 cm upper mineral soil
3 layer. However, it is known that root turnover from trees can incorporate organic matter
4 deeper than 30 cm (Brown and Lugo 1990; Jackson et al. 1996; Jobbágy and Jackson 2000;
5 Trumbore et al. 1995), although C accumulation in root biomass was not taken into account
6 here. The subsoil horizons were very variable (A2, AB, B), and the SOC contents were
7 different, which may have prevented identification of any clear trends.

8 Furthermore, within the same species, the SOC gains were slightly higher in the stands
9 with higher site indices, which reflects the influence of the greater biomass production on
10 litter production. Nevertheless, the effect of the site index was not able to be evaluated
11 accurately, since site index was rather high in most cases. This aspect is of interest, as climate
12 change is expected to change site index, and therefore net primary production, worldwide, and
13 the effect on changes in SOC required further investigation because SOM decomposition is
14 also expected to change, adding a degree of uncertainty as regards the final effect. Soil texture
15 is one of the most important factors controlling SOC dynamics, and SOC increases with clay
16 content in afforested soils (Mendham et al. 2003). The present study, however, did not
17 identify any changes in the SOC dynamics attributable to soil texture, probably because the
18 soils were rather homogeneous as regards this parameter.

19 **Influence of species**

20 The results of the present study revealed important differences in the SOC dynamics
21 following afforestation, attributable to the tree species and the associated vegetation.
22 Although SOC losses were always recorded after afforestation, in the soils under the two
23 *Eucalyptus* species, losses were generally lower and the periods of loss were shorter. In the *P.*
24 *radiata* stands there was clear net gain of SOC from 25 years onwards, because of the longer
25 rotation. In both *Eucalyptus* stands, the compensation ages were close to the end of the
26 rotation, which implies no net gains prior to cutting in most cases. Longer rotation in
27 *Eucalyptus* may lead to positive SOC gains.

28 Moreover, the variability in SOC in the earlier period (0-10 yr) was much higher in both
29 *Eucalyptus* stands than in *P. radiata* (Fig. 2). This was probably due to the death of weeds and
30 herbaceous species in the forest subplot because of shading, which leads to a lower C input to
31 soils in the longer term in all plots, independently of the initial conditions. In the *P. radiata*
32 plots (Table 4) there were no significant differences between the upper and lower part of the
33 CAD curve (Fig. 2).

1 The different patterns in the SOC dynamics may be determined by the different SOM
2 dynamics and litter turnover for the three tree species studied. Thus, it is possible that
3 retention of herbaceous litter in both types of *Eucalyptus* stands prevented SOC losses during
4 the first years after afforestation, as also suggested by Vesterdal et al. (2002), and Huang et al.
5 (2011a). However, the differences in SOC may also be due to the different ground vegetation
6 development in *Eucalyptus* and *P. radiata* stands, which affects the SOC via different
7 mechanisms (Lugo and Brown 1993; Silver et al. 2004). In *P. radiata* plantations, the wider
8 crown intercepting solar radiation exerts a negative influence on the ground vegetation (very
9 large decreases in the ground vegetation occur from the 5th year (Omil et al. 2007)) and
10 probably also negatively affects litter decomposition. Thus, the higher losses of SOC
11 observed in the young *P. radiata* plantations may be due to the lower transfer of organic C to
12 the mineral soil, as a consequence of the lower litter inputs from the ground vegetation and
13 the lower decomposition rate of the litter.

14 The opposite occurs in the *Eucalyptus* plantations, in which the higher crown light-
15 transmission favours higher ground vegetation cover throughout the whole rotation, and with
16 a high presence of grass species (González-Hernández et al. 1998; Silva-Pando et al. 2002).
17 The different ground vegetation cover probably determined the amounts and the type of litter
18 (aerial, root) in these plantations. Thus, the presence of grass in these young plantations
19 resulted in greater belowground C inputs such as root biomass turnover and root exudates
20 (Jones et al. 2009). Grass material is incorporated more rapidly than forest floor material into
21 soil organic matter (Andrade et al. 2008; Laungani and Knops 2009). The lower losses of
22 SOC in *Eucalyptus* soils may therefore be due to the higher inputs of litter from grasses in the
23 ground vegetation, thus compensating for the initial losses of SOC following afforestation.
24 Similar mechanisms has also been suggested by Lemma et al. (2006) and Huang et al. (2011a)
25 to explain the greater SOC gains in soils afforested with *Pinus patula* and *E. nitens*,
26 respectively.

27 The C/N ratios in the mineral soil layer of the mature afforested soils under *P. radiata*
28 increased throughout the rotation. This effect has also been reported in other studies (Giddens
29 et al. 1997; Jug et al. 1999; Smethurst and Sadanandan Nambiar 1995; Ussiri et al. 2006), and
30 is probably due to the increased influence of forest litter on SOM quality throughout the
31 rotation. Thus, the C/N ratio of forest litter was 44.8 for *Eucalyptus*, and 62.6 for *P. radiata*,
32 similar to that reported by McGroody et al. (2004) for broadleaf and coniferous species in
33 senesced litter, whereas the C/N ratio in pasture litter was 22.4. The mineral soil C/N ratio
34 clearly reflects a shift from organic input dominated by grass litter, to forest litter containing

1 greater amounts of recalcitrant biopolymers (resins, waxes, suberin and cutin-derived
2 compounds (Chefetz et al. 2002; Otto and Simpson 2006)). The higher C/N ratio in the
3 afforested soils may also be due to a lower presence of legumes in the understory vegetation
4 (Corbeels et al. 2002) and to higher N immobilization in trees.

5 The C/N ratio did not increase significantly with age in *Eucalyptus* stands, possibly
6 because of the presence of more grass in the underground vegetation in these plantations. Soil
7 analyses revealed the presence of more carbohydrates in the SOM in these mature stands,
8 reflecting different sources of litter (possibly due to the input of root litter and root exudates)
9 relative to the mineral soil under *P. radiata*, in which more recalcitrant compounds were
10 identified (data not shown).

11 **Increasing the C sink capacity by tree species selection and management**

12 The data obtained in the present study show that of the species studied, *E. nitens* has the
13 highest C sink capacity, followed by *E. globulus* and, very closely by *P. radiata*. The mean
14 rates of C sequestration (biomass and soil) estimated in this study for the most common
15 rotations (Table 5) ranged between 8.7 and 12.6 Mg C ha⁻¹ yr⁻¹ (average value for the three
16 species, 10.9 Mg C ha⁻¹ yr⁻¹). Considering that the afforested area in northern Spain using
17 these three species can be estimated as 135,000 ha for the period 1994-2006 (MAPA 2006),
18 afforestation would have resulted in a sink of 1.2-1.7 Tg C yr⁻¹ (average 1.5 Tg C yr⁻¹), with
19 respect to the Spanish CO₂ emissions (101 Tg C in the year 2009, (MMAMRM 2010)). This
20 indicates the significant contribution of afforestation to the mitigation of CO₂ emissions, and
21 also shows that selection of the tree species is a major factor influencing the C sink capacity.

22 Prolongation of the rotation by 10 and 5 years for *Eucalyptus* and *P. radiata*
23 respectively resulted in a C sequestration rate ranging between 10.9 and 14.2 Mg C ha⁻¹ yr⁻¹
24 (average value for the three species, 12.7 Mg C ha⁻¹ yr⁻¹, Table 5), which implies a sink of
25 1.6-2.3 Tg C yr⁻¹ (average 1.7 Tg C ha⁻¹ yr⁻¹). These and previous results (Balboa-Murias et
26 al. 2006; Diaz-Balteiro et al. 2009) show that in order to maximize the C sink capacity,
27 plantations should be managed according to the optimal harvesting schedules for these
28 species. In the present study, the data show that the C sink capacity of these plantations can be
29 increased greatly by prolonging the rotation period.

30 The selection of tree species and the harvest scheduling may also favour C gains in the
31 soil. The contribution of the soil (forest floor plus mineral soil) to the overall C sequestration
32 ranged from 8 to 18 % (in average, 15 %), which is similar to the inputs reported by De Vries

1 et al. (2006) and Woodbury et al. (2006), in Europe and United States, respectively and lower
2 than those reported by Liski et al. (2002).

3 **Conclusions**

4 In this study an intensive sampling scheme was used to assess the C sink capacity of
5 forest stands of the three species most commonly used in afforestation programmes in
6 northern Spain. The high spatial variability in the different compartments illustrates the risk of
7 reaching wrong conclusions about SOC dynamics when the experimental design does not
8 cover most of the variability.

9 The humid temperate climate resulted in C accumulation rates as high as 11-15 Mg C
10 ha⁻¹ in the total system in the first 20 yr, depending on the species and the rotation length. The
11 results of the study show how selection of the tree species is a major factor influencing the
12 post afforestation C sink capacity, affecting the amounts of C accumulated in both biomass
13 and soil.

14 The role of the tree species is particularly important during the first years after
15 afforestation when the litter input from herbaceous vegetation may compensate for losses of
16 SOC. The patterns of SOC dynamics differed greatly in relation to the different tree species
17 used in the afforestation, and were determined by transfer of C to the soil via the roots of the
18 ground vegetation, SOM decomposition, the turnover rate of the forest floor and its
19 macrofauna-mediated incorporation at depth. Both root and forest floor turnover were lower
20 in the *P. radiata* plantations than in the *Eucalyptus* plantations, which may explain the higher
21 SOC losses during the first years after afforestation.

22 The humid temperate climate, along with the lack of physically protected SOM (sandy
23 loam texture of the soils) favoured important losses of SOC in the uppermost mineral soils
24 during the first years.

25 The study provides accurate information on the success of these afforestation
26 programmes as regards CO₂ mitigation. To enhance the C sink capacity, plantations should be
27 managed according to optimal harvesting schedules for the species. Elongation of the rotation
28 length led to larger C sink capacities in all three species. This is especially important in such
29 fast growing plantations, in which harvesting in short rotations may lead to continuous loss of
30 SOC.

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6 References

- 7 Alvarez E, Fernández Marcos ML, Torrado V, Fernández Sanjurjo MJ (2008) Dynamics of macronutrients
8 during the first stages of litter decomposition from forest species in a temperate area (Galicia, NW Spain).
9 *Nutr Cycling Agroecosyst* 80:243-256. doi: 10.1007/s10705-007-9140-4
- 10 Andrade HJ, Brook R, Ibrahim M (2008) Growth, production and carbon sequestration of silvopastoral systems
11 with native timber species in the dry lowlands of Costa Rica. *Plant Soil* 308:11-22. doi: 10.1007/s11104-008-
12 9600-x
- 13 Balboa-Murias MA, Rodríguez-Soalleiro R, Merino A, Álvarez-González JG (2006) Temporal variations and
14 distribution of carbon stocks in aboveground biomass of radiata pine and maritime pine pure stands under
15 different silvicultural alternatives. *For Ecol Manage* 237:29-38. doi: 10.1016/j.foreco.2006.09.024
- 16 Balesdent J, Chenu C, Balabane M (2000) Relationship of soil organic matter dynamics to physical protection
17 and tillage. *Soil Tillage Res* 53:215-230. doi: 10.1016/S0167-1987(99)00107-5
- 18 Bashkin MA, Binkley D (1998) Changes in soil carbon following afforestation in Hawaii. *Ecology* 79:828-833.
19 doi: 10.1890/0012-9658(1998)079[0828:CISCFA]2.0.CO;3B2
- 20 Bell MJ, Harch GR, Bridge BJ (1995) Effects of continuous cultivation on Ferrosols in subtropical southeast
21 Queensland. I. Site characterization, crop yields and soil chemical status. *Aust J Agric Res* 46:237-253. doi:
22 10.1071/AR9950237
- 23 Berg B, Johansson MB, Nilsson A, Gundersen P, Norell L (2009) Sequestration of carbon in the humus layer of
24 Swedish forests—direct measurements. *Can J For Res* 39:962-975. doi: 10.1139/X09-022
- 25 Berg B (2000) Litter decomposition and organic matter turnover in northern forest soils. *For Ecol Manage*
26 133:13-22. doi: 10.1016/S0378-1127(99)00294-7
- 27 Berthrong ST, Jobbágy EG, Jackson RB (2009) A global meta-analysis of soil exchangeable cations, pH, carbon,
28 and nitrogen with afforestation. *Ecol Appl* 19:2228-2241. doi: 10.1890/08-1730.1
- 29 Blake GR, Hartge KH (1986) Bulk density. In: Klute A (ed) *Methods of soil analysis*, 2nd edn. ASA and SSSA,
30 Madison, WI, pp 363-375
- 31 Brown S, Lugo AE (1990) Effects of forest clearing and succession on the carbon and nitrogen content of soils
32 in Puerto Rico and US Virgin Islands. *Plant Soil* 124:53-64. doi: 10.1007/BF00010931
- 33 Carneiro M, Serrão V, Fabião A, Madeira M, Balsemão I, Hilário L (2009) Does harvest residue management
34 influence biomass and nutrient accumulation in understory vegetation of *Eucalyptus globulus* Labill.
35 plantations in a Mediterranean environment?. *For Ecol Manage* 257:527-535. doi:
36 10.1016/j.foreco.2008.09.027
- 37 Cerli C, Celi L, Kaiser K, Guggenberger G, Johansson MB, Cignetti A, Zanini E (2008) Changes in humic
38 substances along an age sequence of Norway spruce stands planted on former agricultural land. *Org Geochem*
39 39:1269-1280. doi: 10.1016/j.orggeochem.2008.06.001

- 1 Chefetz BS, Salloum MJ, Deshmukh AP, Hatcher PG (2002) Structural components of humic acids as
2 determined by chemical modifications and carbon-13 NMR, pyrolysis-, and thermochemolysis-gas
3 chromatography/mass spectrometry. *Soil Sci Soc Am J* 66:1159-1171. doi: 10.2136/sssaj2002.1159
- 4 Conant RT, Smith GR, Paustian K (2003) Spatial variability of soil carbon in forested and cultivated sites:
5 Implications for change detection. *J Environ Qual* 32:278-286. doi: 10.2134/jeq2003.2780
- 6 Conteh A, Lefroy RDB, Blair GJ (1997) Dynamics of organic matter in soil as determined by variations in
7 $^{13}\text{C}/^{12}\text{C}$ isotopic ratios and fractionation by ease of oxidation. *Aust J Soil Res* 35:881-890. doi:
8 10.1071/S96107
- 9 Corbeels M, O'Connell AM, McMurtrie RE, Grove TS, Mendham DS (2002) Modelling changes in nitrogen
10 mineralisation following conversion of improved pasture to eucalypt plantation. *Agronomie* 22:801-815. doi:
11 10.1051/agro:2002059
- 12 Corre MD, Schnabel RR, Shaffer JA (1999) Evaluation of soil organic carbon under forests, cool-season and
13 warm-season grasses in the northeastern US. *Soil Biology and Biochemistry*, 31:1531-1539. doi:
14 10.1016/S0038-0717(99)00074-7
- 15 Covington WW (1981) Changes in forest floor organic matter and nutrient content following clear cutting in
16 northern hardwoods. *Ecology* 62:41-48. doi: 10.2307/1936666
- 17 De Vries WIM, Reinds GJ, Gundersen PER, Sterba H (2006) The impact of nitrogen deposition on carbon
18 sequestration in European forests and forest soils. *Global Change Biol* 12:1151-1173. doi: 10.1111/j.1365-
19 2486.2006.01151.x
- 20 Detwiler RP (1986) Land use change and the global carbon cycle: the role of tropical soils. *Biogeochemistry*
21 2:67-93. doi: 10.1007/BF02186966
- 22 Diaz-Balteiro L, Bertomeu M, Bertomeu M (2009) Optimal harvest scheduling in *Eucalyptus* plantations: A case
23 study in Galicia (Spain). *Forest Policy Econ* 11:548-554. doi: 10.1016/j.forpol.2009.07.005
- 24 Don A, Rebmann C, Kolle O, Scherer-Lorenzen M, Schulze ED (2009) Impact of afforestation-associated
25 management changes on the carbon balance of grassland. *Global Change Biol* 15:1990-2002. doi:
26 10.1111/j.1365-2486.2009.01873.x
- 27 EU (1992) COUNCIL REGULATION (EEC) N° 2080/92 of 30 June 1992 Instituting a Community aid Scheme
28 for Forestry Measures in Agriculture
- 29 Fearnside PM, Barbosa RI (1998) Soil carbon changes from conversion of forest to pasture in Brazilian
30 Amazonia. *For Ecol Manage* 108:147-166. doi: 10.1016/S0378-1127(98)00222-9
- 31 Fernández-Núñez E, Rigueiro-Rodríguez A, Mosquera-Losada MR (2010) Carbon allocation dynamics one
32 decade after afforestation with *Pinus radiata* D. Don and *Betula alba* L. under two stand densities in NW
33 Spain. *Ecol Eng* 35:876-890. doi: 10.1016/j.ecoleng.2010.03.007
- 34 Giddens KM, Parfitt RL, Percival HJ (1997) Comparison of some soil properties under *Pinus radiata* and
35 improved pasture. *N Z J Agric Res* 40:409-416. doi: 10.1080/00288233.1997.9513262
- 36 Glaser B, Turrion MB, Solomon D, Ni A, Zech W (2000) Soil organic matter quantity and quality in mountain
37 soils of the Alay Range, Kyrgyzia, affected by land use change. *Biol Fertility Soils* 31:407-413. doi:
38 10.1007/s003749900187
- 39 Goidts E, Van Wesemael B, Crucifix M (2009) Magnitude and sources of uncertainties in soil organic carbon
40 (SOC) stock assessments at various scales. *Eur J Soil Sci* 60:723-739. doi: 10.1111/j.1365-2389.2009.01157.x

- 1 Golchin A, Oades JM, Skjemstad JO, Clarke P (1994) Soil structure and carbon cycling. *Aust J Soil Res*
2 32:1043-1068. doi: 10.1071/SR9941043
- 3 González-Hernández MP, Silva-Pando FJ, Jiménez MC (1998) Production patterns of understory layers in
4 several Galician (NW Spain) woodlands: Seasonality, net productivity and renewal rates. *For Ecol Manage*
5 109:251-259. doi: 10.1016/S0378-1127(98)00253-9
- 6 Guggenberger G, Zech W (1999) Soil organic matter composition under primary forest, pasture, and secondary
7 forest succession, Region Huetar Norte, Costa Rica. *For Ecol Manage* 124:93-104. doi: 10.1016/S0378-
8 1127(99)00055-9
- 9 Guo LB, Wang M, Gifford RM (2007) The change of soil carbon stocks and fine root dynamics after land use
10 change from a native pasture to a pine plantation. *Plant Soil* 299:251-262. doi: 10.1007/s11104-007-9381-7
- 11 Guo LB, Gifford RM (2002) Soil carbon stocks and land use change: a meta analysis. *Global Change Biol*
12 8:345-360. doi: 10.1046/j.1354-1013.2002.00486.x
- 13 Hobbie SE, Ogdahl M, Chorover J, Chadwick OA, Oleksyn J, Zytzkowiak R, Reich PB (2007) Tree species
14 effects on soil organic matter dynamics: the role of soil cation composition. *Ecosystems* 10:999-1018. doi:
15 10.1007/s10021-007-9073-4
- 16 Hooker TD, Compton JE (2003) Forest ecosystem carbon and nitrogen accumulation during the first century
17 after agricultural abandonment. *Ecol Appl* 13:299-313. doi: 10.1890/1051-
18 0761(2003)013[0299:FECANA]2.0.CO%3B2
- 19 Huang Z, Davis MR, Condrón LM, Clinton PW (2011a) Soil carbon pools, plant biomarkers and mean carbon
20 residence time after afforestation of grassland with three tree species. *Soil Biol Biochem* 43:1341-1349. doi:
21 10.1016/j.soilbio.2011.03.008
- 22 Huang Z, Clinton PW, Davis MR (2011b) Post-harvest residue management effects on recalcitrant carbon pools
23 and plant biomarkers within the soil heavy fraction in *Pinus radiata* plantations. *Soil Biol Biochem* 43:404-
24 412. doi: 10.1016/j.soilbio.2010.11.008
- 25 Huggett RJ (1998) Soil chronosequences, soil development, and soil evolution: a critical review. *Catena* 32:155-
26 172. doi: 10.1016/S0341-8162(98)00053-8
- 27 IPCC (2007) Climate change 2007: Synthesis report. Contribution of Working Groups I, II and III to the Fourth
28 Assessment Report of the Intergovernmental Panel on Climate Change. Core Writing Team, Pachauri, R.K.
29 and Reisinger, A. (Eds.). United Nations Office, Geneva, Switzerland
- 30 IUSS Working Group WRB (2006) World reference base for soil resources 2006. *World Soil Resources Reports*
31 No. 103. FAO, Rome
- 32 Jackson RB, Banner JL, Jobbágy EG, Pockman WT, Wall DH (2002) Ecosystem carbon loss with woody plant
33 invasion of grasslands. *Nature* 418:623-626. doi: 10.1038/nature00910
- 34 Jackson RB, Canadell J, Ehleringer JR, Mooney HA, Sala OE, Schulze ED (1996) A global analysis of root
35 distributions for terrestrial biomes. *Oecologia* 108:389-411. doi: 10.1007/BF00333714
- 36 Jandl R, Lindner M, Vesterdal L, Bauwens B, Baritz R, Hagedorn F, Johnson DW, Minkkinen K, Byrne KA
37 (2007) How strongly can forest management influence soil carbon sequestration?. *Geoderma* 137:253-268.
38 doi: 10.1016/j.geoderma.2006.09.003
- 39 Jobbágy EG, Jackson RB (2000) The vertical distribution of soil organic carbon and its relation to climate and
40 vegetation. *Ecol Appl* 10:423-436. doi: 10.1890/1051-0761(2000)010[0423:TVDOSO]2.0.CO%3B2

- 1 Johnston MH, Homann PS, Engstrom JK, Grigal DF (1996) Changes in ecosystem carbon storage over 40 years
2 on an old-field/forest landscape in east-central Minnesota. For *Ecol Manage* 83:17-26. doi: 10.1016/0378-
3 1127(96)03704-8
- 4 Jones DL, Nguyen C, Finlay RD (2009) Carbon flow in the rhizosphere: carbon trading at the soil–root interface.
5 *Plant Soil* 321:5-33. doi: 10.1007/s11104-009-9925-0
- 6 Jug A, Makeschin F, Rehfuess KE, Hofmann-Schielle C (1999) Short-rotation plantations of balsam poplars,
7 aspen and willows on former arable land in the Federal Republic of Germany. III. Soil ecological effects. For
8 *Ecol Manage* 121:85-99. doi: 10.1016/S0378-1127(98)00558-1
- 9 Kanerva S, Smolander A (2007) Microbial activities in forest floor layers under silver birch, Norway spruce and
10 Scots pine. *Soil Biol Biochem* 39:1459-1467. doi: 10.1016/j.soilbio.2007.01.002
- 11 Kasel S, Singh S, Sanders GJ, Bennett LT (2011) Species-specific effects of native trees on soil organic carbon
12 in biodiverse plantings across north-central Victoria, Australia. *Geoderma* 161:95-106. doi:
13 10.1016/j.geoderma.2010.12.014
- 14 Kirschbaum MUF, Guo LB, Gifford RM (2008) Why does rainfall affect the trend in soil carbon after converting
15 pastures to forests? A possible explanation based on nitrogen dynamics. For *Ecol Manage* 255:2990-3000. doi:
16 10.1016/j.foreco.2008.02.005
- 17 Laganière J, Angers DA, Paré D (2010) Carbon accumulation in agricultural soils after afforestation: a
18 meta - analysis. *Global Change Biol* 16:439-453. doi: 10.1111/j.1365-2486.2009.01930.x
- 19 Laungani R, Knops JMH (2009) The impact of co - occurring tree and grassland species on carbon sequestration
20 and potential biofuel production. *GCB Bioenergy* 1:392-403. doi: 10.1111/j.1757-1707.2009.01031.x
- 21 Leirós MC, Trasar-Cepeda C, Seoane S, Gil-Sotres F (2000) Biochemical properties of acid soils under climax
22 vegetation (Atlantic oakwood) in an area of the European temperate-humid zone (Galicia, NW Spain): general
23 parameters. *Soil Biol Biochem* 32:733-745. doi: 10.1016/S0038-0717(99)00195-9
- 24 Lemma B, Nilsson I, Kleja DB, Olsson M, Knicker H (2007) Decomposition and substrate quality of leaf litters
25 and fine roots from three exotic plantations and a native forest in the southwestern highlands of Ethiopia. *Soil*
26 *Biol Biochem* 39:2317-2328. doi: 10.1016/j.soilbio.2007.03.032
- 27 Lemma B, Kleja DB, Nilsson I, Olsson M (2006) Soil carbon sequestration under different exotic tree species in
28 the southwestern highlands of Ethiopia. *Geoderma* 136:886-898. doi: 10.1016/j.geoderma.2006.06.008
- 29 Liski J, Perruchoud D, Karjalainen T (2002) Increasing carbon stocks in the forest soils of western Europe. For
30 *Ecol Manage* 169:159-175. doi: 10.1016/S0378-1127(02)00306-7
- 31 Lugo AE, Brown S (1993) Management of tropical soils as sinks or sources of atmospheric carbon. *Plant Soil*
32 149:27-41. doi: 10.1007/BF00010760
- 33 Maillard ÉP, Munson D, Alison D (2010) Soil carbon stocks and carbon stability in a twenty-year-old temperate
34 plantation. *Soil Sci Soc Am J* 74:1775-1785. doi: 10.2136/sssaj2009.0273
- 35 MAPA (2006) Forestación de Tierras Agrícolas: Análisis de su evolución y contribución a la fijación de carbono
36 y al uso racional de la tierra. Spanish Ministry of Agriculture, Fisheries and Food, Madrid (Spain)
- 37 Marín-Spiotta E, Silver WL, Swanston CW, Ostertag R (2009) Soil organic matter dynamics during 80 years of
38 reforestation of tropical pastures. *Global Change Biol* 15:1584-1597. doi: 10.1111/j.1365-2486.2008.01805.x

- 1 Martius C, Höfer H, Garcia MVB, Römbke J, Förster B, Hanagarth W (2004a) Microclimate in agroforestry
2 systems in central Amazonia: does canopy closure matter to soil organisms?. *Agrofor Syst* 60:291-304. doi:
3 10.1023/B:AGFO.0000024419.20709.6c
- 4 Martius C, Höfer H, Garcia MVB, Römbke J, Hanagarth W (2004b) Litter fall, litter stocks and decomposition
5 rates in rainforest and agroforestry sites in central Amazonia. *Nutr Cycling Agroecosyst* 68:137-154. doi:
6 10.1023/B:FRES.0000017468.76807.50
- 7 McGroddy ME, Daufresne T, Hedin L (2004) Sacling of C:N:P stoichiometry in forests worldwide: implications
8 of terrestrial redfield-type ratios. *Ecology* 85:2390-2401. doi: 10.1890/03-0351
- 9 Mendham DS, O'Connell AM, Grove TS (2003) Change in soil carbon after land clearing or afforestation in
10 highly weathered lateritic and sandy soils of south-western Australia. *Agriculture, Ecosystems & Environment*
11 95:143-156. doi: 10.1016/S0167-8809(02)00105-6
- 12 Merino A, Balboa MA, Rodríguez-Soalleiro R, González JG (2005) Nutrient exports under different harvesting
13 regimes in fast-growing forest plantations in southern Europe. *For Ecol Manage* 207:325-339. doi:
14 10.1016/j.foreco.2004.10.074
- 15 Merino A, Fernández-López A, Solla-Gullón F, Edeso JM (2004) Soil changes and tree growth in intensively
16 managed *Pinus radiata* in northern Spain. *For Ecol Manage* 196:393-404. doi: 10.1016/j.foreco.2004.04.002
- 17 MMAMRM (2010) Inventario de gases de efecto invernadero de España. Sumario de resultados. Ministerio de
18 Medio Ambiente, Medio Rural y Marino. Secretaría de Estado de Cambio Climático, Madrid (Spain)
- 19 Morris SJ, Bohm S, Haile - Mariam S, Paul EA (2007) Evaluation of carbon accrual in afforested agricultural
20 soils. *Global Change Biol* 13:1145-1156. doi: 10.1111/j.1365-2486.2007.01359.x
- 21 Murty D, Kirschbaum MUF, Mcmurtrie RE, Mcgilvray H (2002) Does conversion of forest to agricultural land
22 change soil carbon and nitrogen? A review of the literature. *Global Change Biol* 8:105-123. doi:
23 10.1046/j.1354-1013.2001.00459.x
- 24 Neill C, Melillo JM, Steudler PA, Cerri CC, de Moraes JFL, Piccolo MC, Brito M (1997) Soil carbon and
25 nitrogen stocks following forest clearing for pasture in the southwestern Brazilian Amazon. *Ecol Appl* 7:1216-
26 1225. doi: 10.1890/1051-0761(1997)007[1216:SCANSF]2.0.CO%3B2
- 27 Nilsson S, Schopfhauser W (1995) The carbon-sequestration potential of a global afforestation program. *Clim*
28 *Change* 30:267-293. doi: 10.1007/BF01091928
- 29 Omil B, Mosquera-Losada R, Merino A (2007) Responses of a Non N-Limited Forest Plantation to the
30 Application of Alkaline-Stabilized Dewatered Dairy Factory Sludge. *J Environ Qual* 36:1765-1774. doi:
31 10.2134/jeq2007.0057
- 32 Ostertag R, Marín-Spiotta E, Silver WL, Schulten J (2008) Litterfall and decomposition in relation to soil carbon
33 pools along a secondary forest chronosequence in Puerto Rico. *Ecosystems* 11:701-714. doi: 10.1007/s10021-
34 008-9152-1
- 35 Otto A, Simpson MJ (2006) Sources and composition of hydrolysable aliphatic lipids and phenols in soils from
36 western Canada. *Org Geochem* 37:385-407. doi: 10.1016/j.orggeochem.2005.12.011
- 37 Paul KI, Polglase PJ (2004) Prediction of decomposition of litter under eucalypts and pines using the FullCAM
38 model. *For Ecol Manage* 191:73-92. doi: 10.1016/j.foreco.2003.11.007
- 39 Paul KI, Polglase PJ, Nyakuengama JG, Khanna PK (2002) Change in soil carbon following afforestation. *For*
40 *Ecol Manage* 168:241-257. doi: 10.1016/S0378-1127(01)00740-X

1 Paustian K, Six J, Elliott ET, Hunt HW (2000) Management options for reducing CO₂ emissions from
2 agricultural soils. *Biogeochemistry* 48:147-163. doi: 10.1023/A:1006271331703

3 Pérez-Cruzado C, Merino A, Rodríguez-Soalleiro R (2011) A management tool for estimating bioenergy
4 production and carbon sequestration in *Eucalyptus globulus* and *Eucalyptus nitens* grown as short rotation
5 woody crops in north-west Spain. *Biomass Bioenerg*:(in press). doi: 10.1016/j.biombioe.2011.03.020

6 Pérez-Cruzado C, Rodríguez Soalleiro R (2011) Improvement in accuracy of aboveground biomass estimation in
7 *Eucalyptus nitens* plantations: effect of bole sampling intensity and explanatory variables. *For Ecol Manage*
8 261:2016-2028. doi: 10.1016/j.foreco.2011.02.028

9 Pérez-Cruzado C, Solla-Gullón F, Merino A, Rodríguez-Soalleiro R (2011) Analysis of growth and nutrition of a
10 young *Castanea×coudercii* plantation after application of wood-bark ash. *Eur J Forest Res* 130:209-217. doi:
11 10.1007/s10342-010-0422-z

12 Poeplau C, Don A, Vesterdal L, Leifeld J, van Wesemael B, Schumacher J, Gensior A (2011) Temporal
13 dynamics of soil organic carbon after land - use change in the temperate zone—carbon response functions as a
14 model approach. *Global Change Biol* in press. doi: 10.1111/j.1365-2486.2011.02408.x

15 Post WM, Kwon KC (2000) Soil carbon sequestration and land - use change: processes and potential. *Global*
16 *Change Biol* 6:317-327. doi: 10.1046/j.1365-2486.2000.00308.x

17 Prescott CE (2010) Litter decomposition: what controls it and how can we alter it to sequester more carbon in
18 forest soils?. *Biogeochemistry* 101:1-17. doi: 10.1007/s10533-010-9439-0

19 Reiners WA, Bouwman AF, Parsons WFJ, Keller M (1994) Tropical rain forest conversion to pasture: changes
20 in vegetation and soil properties. *Ecol Appl* 4:363-377. doi: 10.2307/1941940

21 Rhoades CC, Eckert GE, Coleman DC (2000) Soil carbon differences among forest, agriculture, and secondary
22 vegetation in lower montane Ecuador. *Ecol Appl* 10:497-505. doi: 10.1890/1051-
23 0761(2000)010[0497:SCDAFA]2.0.CO%3B2

24 Richards FJ (1959) A flexible growth function for empirical use. *J Exp Bot* 10:290-301. doi:
25 10.1093/jxb/10.2.290

26 Richter DD, Markewitz D, Trumbore SE, Wells CG (1999) Rapid accumulation and turnover of soil carbon in a
27 re-establishing forest. *Nature* 400:56-58. doi: 10.1038/21867

28 Ross DJ, Tate KR, Scott NA, Feltham CW (1999) Land-use change: effects on soil carbon, nitrogen and
29 phosphorus pools and fluxes in three adjacent ecosystems. *Soil Biol Biochem* 31:803-813. doi:
30 10.1016/S0038-0717(98)00180-1

31 SAS Institute Inc (2004) SAS/STAT 9.1 User's Guide, Cary, N.C.

32 Schimel DS (1995) Terrestrial ecosystems and the carbon cycle. *Global Change Biol* 1:77-91. doi:
33 10.1111/j.1365-2486.1995.tb00008.x

34 Schlesinger WH (1986) Changes in soil carbon storage and associated properties with disturbance and recovery.
35 In: Trabalka JR, Reichle DE (eds) *The changing carbon cycle: a global analysis*. Springer-Verlag, New York
36 (USA), pp 194-220

37 Silva-Pando FJ, González-Hernández MP, Rozados-Lorenzo MJ (2002) Pasture production in a silvopastoral
38 system in relation with microclimate variables in the Atlantic coast of Spain. *Agrofor Syst* 56:203-211. doi:
39 10.1023/A:1021359817311

1 Silver WL, Kueppers LM, Lugo AE, Ostertag R, Matzek V (2004) Carbon sequestration and plant community
2 dynamics following reforestation of tropical pasture. *Ecol Appl* 14:1115-1127. doi: 10.1890/03-5123
3 Smethurst PJ, Sadanandan Nambiar EK (1995) Changes in soil carbon and nitrogen during the establishment of
4 a second crop of *Pinus radiata* . *For Ecol Manage* 73:145-155. doi: 10.1016/0378-1127(94)03491-E
5 Smith TM, Cramer WP, Dixon RK, Leemans R, Neilson RP, Solomon AM (1993) The global terrestrial carbon
6 cycle. *Wat Air Soil Pollut* 70:19-37. doi: 10.1007/BF01104986
7 Stanturf JA, Madsen P (2002) Restoration concepts for temperate and boreal forests of North America and
8 Western Europe. *Plant Biosyst* 136:143-158. doi: 10.1080/11263500212331351049
9 Stendahl J, Johansson MB, Eriksson E, Nilsson Å, Langvall O (2010) Soil organic carbon in Swedish spruce and
10 pine forests–Differences in stock levels and regional patterns. *Silva Fenn* 44:5-21
11 Thuille A, Schulze ED (2006) Carbon dynamics in successional and afforested spruce stands in Thuringia and
12 the Alps. *Global Change Biol* 12:325-342. doi: 10.1111/j.1365-2486.2005.01078.x
13 Toriyama J, Kato T, Siregar CA, Siringoringo HH, Ohta S, Kiyono Y (2011) Comparison of depth- and mass-
14 based approaches for estimating changes in forest soil carbon stocks: A case study in young plantations and
15 secondary forests in West Java, Indonesia. *For Ecol Manage* 262:1659-1667. doi:
16 10.1016/j.foreco.2011.07.027
17 Trumbore SE, Davidson EA, de Camargo PB, Nepstad DC, Martinelli LA (1995) Belowground cycling of
18 carbon in forests and pastures of Eastern Amazonia. *Global Biogeochem Cycles* 9:515-528. doi:
19 10.1029/95GB02148
20 Turner J, Lambert M (2000) Change in organic carbon in forest plantation soils in eastern Australia. *For Ecol*
21 *Manage* 133:231-247. doi: 10.1016/S0378-1127(99)00236-4
22 UNFCCC (2002) Views from parties on issues related to modalities for the inclusion of afforestation and
23 reforestation project activities under the Clean Development Mechanism in the first commitment period.
24 Submissions for parties. United Nations Framework Convention on Climate Change, United Nations Office,
25 Geneva
26 Ussiri D, Lal P, Jacinthe PA (2006) Soil properties and carbon sequestration of afforested pastures in reclaimed
27 minesoils of Ohio. *Soil Sci Soc Am J* 70:1797-1806. doi: 10.2136/sssaj2005.0352
28 Veldkamp E (1994) Organic carbon turnover in three tropical soils under pasture after deforestation. *Soil Sci Soc*
29 *Am J* 58:175-180. doi: 10.2136/sssaj1994.03615995005800010025x
30 Vesterdal L, Schmidt IK, Callesen I, Nilsson LO, Gundersen P (2008) Carbon and nitrogen in forest floor and
31 mineral soil under six common European tree species. *For Ecol Manage* 255:35-48. doi:
32 10.1016/j.foreco.2007.08.015
33 Vesterdal L, Ritter E, Gundersen P (2002) Change in soil organic carbon following afforestation of former arable
34 land. *For Ecol Manage* 169:137-147. doi: 10.1016/S0378-1127(02)00304-3
35 Vitorello VA, Cerri CC, Andreux F, Feller C, Victoria RL (1989) Organic matter and natural carbon-13
36 distribution in forested and cultivated oxisols. *Soil Sci Soc Am J* 53:773-778. doi:
37 10.2136/sssaj1989.03615995005300030024x
38 von Lützow M, Kogel-Knabner I, Ekschmitt K, Matzner E, Guggenberger G, Marschner B, Flessa H (2006)
39 Stabilization of organic matter in temperate soils: mechanisms and their relevance under different soil
40 conditions-a review. *Eur J Soil Sci* 57:426-445. doi: 10.1111/j.1365-2389.2006.00809.x

1 Walker SM, Desanker PV (2004) The impact of land use on soil carbon in Miombo Woodlands of Malawi. For
2 Ecol Manage 203:345-360. doi: 10.1016/j.foreco.2004.08.004
3 Woodbury PB, Heath LS, Smith JE (2006) Land use change effects on forest carbon cycling throughout the
4 southern United States. J Environ Qual 35:1348-1363. doi: 10.2134/jeq2005.0148
5 Zak DR, Grigal DF, Gleeson S, Tilman D (1990) Carbon and nitrogen cycling during old-field succession:
6 constraints on plant and microbial biomass. Biogeochemistry 11:111-129. doi: 10.1007/BF00002062
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18 **FIGURE LEGENDS**

19 Fig. 1. Changes in total aboveground biomass C throughout the rotation. Continuous lines indicate the fitted
20 Richards model; dotted lines are 95% confidence levels for the mean. *E. nitens* red; *E. globulus* green; *P. radiata*
21 black.

22 Fig. 2. Changes in carbon density accumulation (Mg ha^{-1}) in the forest floor and mineral soils throughout the
23 first rotation after afforestation. Dark shaded area: 95% confidence limits for 0-15 cm depth mineral soil (CAD);
24 light shaded area: 95% confidence limits for the forest floor (n= 40 for each of the three species).

25 Fig. 3. Changes in soil carbon density with time since afforestation for each soil depth. (n=40 for each species).

26 Fig. 4. Changes in total soil C (forest floor plus 0-30 cm mineral soil) throughout the rotation for all three species
27 considered.

28 Fig. 5. Changes in the C density accumulation rate ($\text{Mg ha}^{-1} \text{ yr}^{-1}$) in aboveground biomass, forest floor and
29 mineral soil (up to 15 cm depth) over time (n= 40 for each of the three species).
30

1 **TABLES**

2 Table 1. Main site characteristics (average and standard deviation) of the stands studied.

		<i>E. globulus</i>	<i>E. nitens</i>	<i>P. radiata</i>
Number of stands		40	40	40
Age interval	(yr)	1-23	2-18	2-40
Stand density	(trees ha ⁻¹)	1108 (309)	1011 (258)	1146 (410)
Bedrock (Granitic rock /Schist-Slates)	(n)	8 /32	5/35	25/15
Site Index*	(m)	23.3 (6.5)	15.3 (4.4)	24.8 (4.4)
Altitude	(m)	242 (173)	517 (63)	466 (115)
Average annual temperature	(°C)	13.3 (1.0)	11.6 (0.5)	11.5 (0.8)
Accumulated annual precipitation	(mm)	1488 (377)	1434 (322)	1213 (219)

3 *Reference ages for site index were 10, 6 and 20 years for *E. globulus*, *E. nitens* and *P. radiata* respectively.

4

5 Table 2. Aboveground biomass carbon model parameters and fits for age since afforestation.

Specie	Parameter estimate (Std. error)			RMSE	Adjust. R ²
	<i>b</i> ₀	<i>b</i> ₁	<i>b</i> ₂		
<i>E. globulus</i>	520.3 (6.23)	0.0589 (0.0021)	2.356 (0.0325)	3.0331	0.718
<i>E. nitens</i>	784.1 (10.41)	0.0393 (0.0013)	1.815 (0.0218)	2.1483	0.835
<i>P. radiata</i>	1569.5 (13.75)	0.0126 (0.0008)	1.466 (0.0201)	4.3544	0.788

6

7 Table 3. Average values (and standard deviations) of relative difference in carbon density (CRD, %) considering
8 all species jointly (n= 120), and each species separately (n=40). Significant differences for a given soil depth are
9 indicated by different letters.

	<i>T</i> (years)	Mineral soil layers (cm)				
		0-5	5-15	15-30	0-15	0-30
<i>E. globulus</i>	0-5	-4.1 (17.8)	12.3 (43.6)	28.4 (46.4)	-0.1 (22.0)	13.4 (37.6)
	6-10	-9.4 (34.8)	-9.0 (45.1)	-4.1 (57.9)	-9.2 (39.6)	-8.4 (43.6)
	11-15	5.4 (45.4)	19.2 (110.4)	4.7 (84.2)	6.8 (66.5)	6.9 (72.7)
	16-20	8.7 (30.0)	-0.1 (38.7)	-2.5 (42.8)	3.2 (24.3)	-1.4 (30.2)
<i>E. nitens</i>	0-5	-3.1 (18.5)	13 (34.8)	19.6 (37.4)	-0.3 (24.6)	8.0 (27.9)
	6-10	-6.0 (29.8)	3.4 (41.7)	-17.3 (46.8)	-1.4 (26.1)	-8.6 (31.9)
	11-15	-8.9 (22.4)	-6.3 (29.5)	5.0 (62.8)	-8.9 (21.6)	-4.5 (33.7)
	16-20	29.9 (71.6)	-12.3 (13.8)	-28.2 (20.2)	6.6 (39.9)	-7.4 (26.0)
<i>P. radiata</i>	0-5	-51.3 (18.6) a	-52.8 (16.5) a	-5.3 (29.5)	-52.0 (16.7) a	-34.1 (17.3) a
	6-10	-23.6 (29.4) ab	-23.4 (13.1) ab	-4.5 (18.8)	-24.1 (10.6) ab	-17.1 (9.0) ab
	11-15	-14.3 (41.8) ab	-16.8 (30.9) ab	-13.8 (31.0)	-15.6 (32.6) ab	-16.0 (29.7) ab
	16-20	32.5 (65.2) bc	-4.4 (28.8) bc	4.6 (47.3)	4.6 (31.4) bc	3.7 (32.6) ab
	>21	28.4 (29.3) bc	13.4 (28.7) bc	10.1 (50.5)	18.9 (23.2) cd	14.7 (27.8) b

1

2 Table 4. Average values (and standard deviations) for soil texture, site index and carbon concentration for the
 3 upper and lower part of the carbon density accumulation curve (Fig. 2). Different letters indicate significant
 4 differences ($p < 0.01$)

	Position	Texture		Site	Initial carbon concentration (%)		
		Clay (%)	Sand (%)	Index (m)	0-5 cm	5-15 cm	15-30 cm
<i>E. globulus</i>	Upper	15.37 (5.3)	42.77 (14.2)	23.6 (7.1)	4.95 (2.2)a	2.92 (1.5)a	2.53 (1.2)a
	Lower	12.11 (5.3)	50.92 (11.1)	23.1 (6.0)	6.67 (2.1)b	5.08 (1.7)b	4.09 (1.4)b
<i>E. nitens</i>	Upper	11.74 (5.7)	44.82 (12.7)	16.1 (4.2)	5.51 (2.8)	3.58 (1.4)a	2.88 (1.2)a
	Lower	12.56 (6.7)	44.98 (18.6)	14.4 (4.6)	6.86 (2.3)	5.62 (2.3)b	4.31 (2.0)b
<i>P. radiata</i>	Upper	8.48 (3.8)	57.19 (10.5)	25.4 (4.5)	3.95 (1.0)	3.13 (0.8)	2.43 (0.5)
	Lower	10.77 (5.4)	50.40 (13.7)	24.3 (4.3)	4.12 (1.1)	3.35 (0.5)	2.37 (0.6)

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6

7 Table 5. C accumulation in aboveground biomass, forest floor and mineral soil (0-15 cm) considering three
 8 possible rotation lengths.

	C pool	<i>E. globulus</i>			<i>E. nitens</i>			<i>P. radiata</i>	
		10 yr ₁	15 yr ₂	20 yr ₃	10 yr ₁	15 yr ₂	20 yr ₃	30 yr _{1,2}	35 yr ₃
C accumulation rate (Mg ha ⁻¹ yr ⁻¹)	Biomass	7.73	9.87	10.93	10.19	12.04	13.00	9.61	9.89
	Forest floor	1.18	0.90	0.87	2.92	2.17	1.51	1.62	1.49
	Mineral soil	-0.24	0.17	0.18	-0.56	-0.05	0.05	0.28	0.27
	Total	8.67	10.94	11.98	12.55	14.16	14.56	11.51	11.65
C accumulation (Mg ha ⁻¹)	Biomass	77.3	148.0	218.6	101.9	180.6	260.0	288.3	346.1
	Forest floor	11.8	13.5	17.3	29.2	32.5	30.2	48.5	52.3
	Mineral soil	-2.4	2.6	3.6	-5.6	-0.7	1.0	8.4	9.5
	Total	86.7	164.1	239.5	125.5	212.4	291.2	345.2	407.9

9

1,2 Most common rotation lengths in afforested agricultural land in the region; 3 Long rotation

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1 Table 6. Changes in upper mineral soil carbon after land use change from agriculture to pasture to forest reported in several reference studies. For the values obtained in the
 2 present study: short term and long term effects show the average and the range of values provided by LOESS analysis. The range of compensation age (*Pinus radiata*) and the
 3 CRD compensation age are also shown.

Source	Species	Reference land use	Forest type	Approach	Mineral soil dept (cm)	SHORT TERM EFFECT		COMP. AGE	LONG TERM EFFECT	
						age (years)	effect	(years)	age (years)	effect
Zak et al. (1990)	<i>Quercus ellipsoidalis</i>	A	S	CH	0-10	8-9	-31%	20	40	+35%
Johnston et al. (1996)	Five forest types	A	S	CH	0-10	5-10	-10%	20	40	+40%
Giddens et al. (1997)	<i>Pinus radiata</i>	P	PL	PP	0-10	-	-	16-24 ₂	-	-
Bashkin & Binkley (1998)	<i>Eucalyptus saligna</i>	A	PL	PP	0-10	-	-	10-13	-	-
Richter et al. (1999)	<i>Pinus taeda</i>	A	PL	LT	0-7.5	6-10	-18%	16-18	35	+22%
Ross et al. (1999)	<i>Pinus radiata</i>	P	PL	PP	0-10	19	-13%	-	-	-
Jug et al. (1999) ₁	<i>Populus</i> spp; <i>Salix viminalis</i>	A, P	PL	LT	0-5	7-10	+50% - +17%	-	-	-
Turner & Lambert (2000)	<i>Eucalyptus grandis</i>	P	PL	PP, CH	0-10	15	-40%	-	35	-40%
Vesterdal et al. (2002)	<i>Quercus robur</i> ; <i>Picea abies</i>	A	PL	CH	0-25	5-10	-15%	-	29	-28%
Hooker & Compton (2003)	<i>Pinus strobus</i>	P	S	CH	0-20	-	-	-	115	0%
Ussiri et al. (2006)	<i>Casuarina</i> spp. <i>Robinia pseudoacacia</i>	P	PL	PP	0-10	10	-16%	-	-	-
Thuille & Schulze (2006)	<i>Picea abies</i>	P	PL	CH	0-50	15-60	-30%	80 ₃	112-93	+0.24- +0.34 MgC ha ⁻¹ yr ⁻¹
Morris et al.(2007)	Coniferous spp. Deciduous spp.	A A	PL PL	PP PP	100 100	- -	- -	- -	65 65	+24.7% +35.6%

This study	<i>Eucalyptus globulus</i>	P	PL	CH	0-15	8	-22.2%	12 ₄	+10.8%
							(-52.0%--+2.7%)	(-15.7%--+21.7%)	21 (+2.0%--+24.2%)
	<i>Eucalyptus nitens</i>	P	PL	CH	0-15	11	-16.7%	16 ₄	+2.2%
							(-34.7%--+0.2%)	(-15.5%--+18.8%)	18 (-15.6%--+18.8%)
	<i>Pinus radiata</i>	P	PL	CH	0-15	5-13	-26.0%	17 (14-25) ₅	+23.5%
							(-34.4%--+17.0%)	(-21.4%--+27.9%)	34 (+10.6%--+37.4%)

1 Where: A, agriculture; P, pasture; S, secondary succession, PL, plantation; CH, chronosequence; PP, paired plots; LT, long term study.

2 ₁ Fertilized short rotation plantations.

3 ₂ For 60% of the studied sites.

4 ₃ Obtained from simulation model.

5 ₄ Compensation age for the average value;

6 ₅ Compensation age for the 95% confidence interval