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Soil CO_2 effluxes, soil carbon balance, and early tree growth following savannah afforestation in Congo: Comparison of two site preparation treatments

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Abstract

Eucalyptus plantations have been introduced since 1978 on savannah soils of the coastal plains of Congo, but there is still little information on the effect of silvicultural practices on soil organic carbon dynamics after afforestation on these savannahs. The objectives of this study were to assess the effects of two experimental site preparation treatments on soil CO_2 efflux, tree growth and soil carbon balance during the first year following plantation establishment. One treatment involved mechanical soil disturbance with disk harrowing (D), whereas in the second treatment (H), savannah grasses were killed by herbicide application before planting, without mechanical soil disturbance. Soil respiration and soil water content were monitored for 1 year following treatment application, at 2-week intervals. We hypothesized that mechanical soil disturbance would increase soil CO_2 efflux, but the results did not support this hypothesis. The cumulated soil CO_2 efflux over 1 year was not significantly different in the two treatments and averaged 658 g C m^{-2}. In contrast, tree growth was significantly increased by disk harrowing, maybe as a result of decreased soil penetration resistance. Carbon inputs to the soil from savannah residues (428 g C m^{-2}) were outweighed by the annual carbon outputs through heterotrophic respiration (505 and 456 g C m^{-2} in the H and D treatments, respectively) leading to a slightly negative soil carbon budget in both treatments 1 year after afforestation.

Keywords: Soil CO_2 efflux; Soil water content; Soil disturbance; Savannah afforestation; Heterotrophic soil respiration; Soil carbon balance

1. Introduction

The area of forest plantations is rapidly increasing throughout the world, especially in the tropics (Brown et al., 1997; FAO, 2003). How these land-use changes will impact on soil fertility, water resources, and on the regional and global carbon cycles are therefore important issues (Garcia-Quijano et al., 2005; Jackson et al., 2005; Laclau et al., 2005). In particular, assessing the changes in soil carbon stocks associated with afforestation is of major interest, since soil organic matter (SOM) is known as an important component of soil fertility through its contribution to soil physical, chemical, and biological properties that affect major ecosystem processes (Fisher and Binkley, 2000; Diaz-Zorita et al., 2002; Lal, 2004). This information is also required to quantify amounts of C sequestrated by afforestation (Paul et al., 2002; Resh et al., 2002; Niu and Duiker, 2006). Afforestation may contribute to the goals of the Kyoto protocol by sequestrating C, and/or producing biomass for energy, thus substituting fossil energy (Deckmyn et al., 2004; Jandl et al., 2007).

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In their reviews of the global literature, Paul et al. (2002), and Guo and Gifford (2002) showed that the most important factors affecting the direction and the magnitude of changes in soil carbon stocks after afforestation were previous land use, soil texture, climate, the type of established forest (broadleaf vs. coniferous species), forest management, and the time since afforestation. An initial decrease in soil organic carbon (SOC) is commonly observed after afforestation, due to the loss of C through decomposition, and the cessation of litter inputs from the previous ecosystem (Trouvé, 1992; Paul et al., 2002, 2003). This initial decrease in soil C is generally followed by a gradual increase, as the lack of litter inputs from the previous ecosystem is progressively compensated by litter inputs from the newly established forest plantation (Trouvé et al., 1994; Paul et al., 2002; Turner et al., 2005). After several years, a new soil C equilibrium is eventually reached, when carbon outflows from decomposition are balanced by carbon inflows from litter production (Batjes and Sombroek, 1997; Guo and Gifford, 2002; Jandl et al., 2007).

Site preparation practices at the time of plantation establishment, such as mechanical disturbance (e.g. disking), fertilization, or residue management (residue removal or prescribed burning) may have significant effects on the magnitude of the initial decrease in soil C, and on the subsequent soil C and fertility dynamics (Johnson, 1992; Turner and Lambert, 2000; Yanai et al., 2003). Soil mechanical disturbance such as ploughing, ripping or disking may enhance early tree growth by limiting the re-establishment of competing vegetation (Nambiar and Sand, 1993), but it may also increase soil C loss by several mechanisms (Post and Kwon, 2000; McLaughlin et al., 2000; Bernoux et al., 2006): (a) by disrupting soil aggregates which protect soil organic matter from decomposition; (b) by stimulating short-term microbial activity through enhanced aeration and (c) by mixing fresh residues into the soil. Soil mechanical disturbance may also impact on soil C dynamics by modifying the soil surface microclimate, e.g. soil water content and temperature (Mallik and Hu, 1997; Pérez-Batallon et al., 2001).

Enhanced soil CO_2 effluxes after soil mechanical disturbance have been demonstrated for a variety of soil and climatic conditions from short-term (a few days; La Scala et al., 2005) or medium-term (a few months; Mallik and Hu, 1997) monitoring of soil respiration. Soil disturbance effects are expected to be strongly dependent on the climate, and on soil texture and initial soil carbon stocks (Post and Kwon, 2000; Six et al., 2002). But few studies have assessed the soil disturbance effect on soil CO_2 efflux in sandy tropical soils (Feller and Beare, 1997).

In the Republic of Congo, 40,000 ha of clonal eucalypt plantations have been established since 1978 on savannah soils of the coastal plains (Trouvé et al., 1994; Laclau et al., 2003a,b), mainly for pulpwood production. The mean productivity of these plantations, throughout 7-year rotations, is about 20 m^3 ha^{-1} $year^{-1}$, despite poor soils with low water-retention capacity, and low soil C and nutrient contents. Several studies have been carried out in order to investigate the sustainability of these plantations (Laclau et al., 2003a,b, 2005), and their impacts on biodiversity (Loumeto and Huttel, 1997). However, the impact of contrasted silvicultural practices on nutrient cycles and on carbon sequestration is still poorly documented (Nzila et al., 2002). The effects of site preparation practices on soil C dynamics after plantation establishment have never been investigated.

The main objectives of the present study were to assess the effects of two experimental site preparation treatments on soil CO_2 efflux, tree growth and soil carbon balance during the first year following afforestation in Congo. One treatment involved soil mechanical disturbance (by disk harrowing), whereas in the second treatment, savannah grasses were killed chemically (using herbicide) before tree planting, without soil mechanical disturbance. Soil respiration and soil water content were monitored for 1 year after treatment application. We hypothesized that mechanical soil disturbance would increase soil CO_2 efflux after afforestation, through enhanced soil organic matter mineralization.

2. Materials and methods

2.1. Study site

Eucalyptus plantations cover about 400 km^2 along the Atlantic coast in the Pointe Noire region in the Republic of Congo. The mean annual air humidity and air temperature are high (85% and 25 °C, respectively) with low seasonal variations (about 2% and 5 °C, respectively). Annual precipitation (1998–2005) averaged 1274 mm with a dry season between May and September. Annual rainfall during the experiment (from April 1, 2005 to April 1, 2006) was about 1216 mm. Rainfall distribution, and other meteorological conditions (air temperature and vapour pressure deficit) recorded at a nearby site (≈5 km from our experimental site) from January 2005 to May 2006 are shown in Fig. 1.

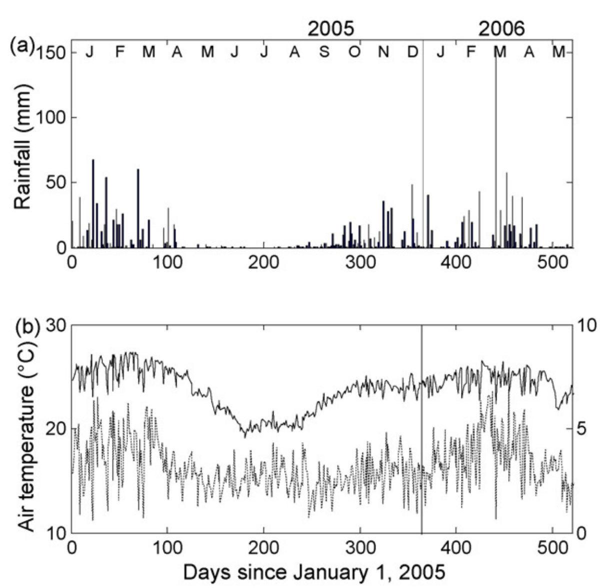

Fig. 1. (a) Daily rainfall and (b) daily mean air temperature (upper line), and vapour pressure deficit (lower line) recorded at the Kissoko site from January 2005 to May 2006.

The experimental site was located on a plateau close to Kissoko village (4°44'41''S, 12°01'16''E, 100 m elevation). The soil was a deep ferralic arenosol (FAO classification) laying on a geological bedrock composed of thick detritic layers of continental origin dated from plio-pleistocene (Jamet and Rieffel, 1976; Trouvé, 1992; Laclau, 2001; Poirier et al., 2002). Mean sand, silt and clay fractions in the 0–50 cm superficial soil layer were about 92, 5 and 3%, respectively (Table 1). This soil was characterised by low water retention, low exchangeable cation contents, and low carbon contents (Table 1). Native vegetation was a savannah dominated by the Poaceae *Loudetia arundinacea* (Hochst.) Steud. and *Hyparrhenia diplandra* (Hack.) (Laclau et al., 2002).

In April 2005, a savannah area of about 0.8 ha was planted with the *Eucalyptus* clone PF1 1-41 at a stocking of 800 trees per hectare (tree spacing: 3.7 m × 3.4 m), which is the current (and nearby optimum) density applied by the forest company EFC (*Eucalyptus* Fiber Congo SA). This clone was the most extensively planted one decade ago (see Bouillet et al., 2002 for details on this clone). The experimental area was divided into three blocks surrounded by a 7 m wide planted buffer strip. In each block, two site preparation treatments were applied before planting. The first treatment involved soil mechanical disturbance, since savannah grasses were killed mechanically by disk harrowing (D treatment). In the second treatment, savannah grasses were killed chemically by herbicide (glyphosate) application, without soil mechanical disturbance (H treatment).

Each block included one 11.1 m wide strip prepared with the D treatment, and one 11.1 m wide strip prepared with the H treatment. Each 11.1 m wide strip was 80 m long and was divided into three 3.7 m wide sub-strips, each including a row of trees after planting. The central 3.7 m sub-strip was used for soil respiration and tree growth measurements, whereas the two external sub-strips were used as buffer strips (Fig. 2). Altogether, six 3.7 m × 80 m plots (2 treatments × 3 blocks) were studied.

Site preparation treatments were applied in March 2005, and soil respiration measurements started about 1 week later on April 1, 2005, and were made during 1 year (till April 19, 2006). Tree planting occurred during the first week of April 2005. After planting, potential re-growth of savannah grasses in the six studied plots was prevented by occasional local applications of glyphosate.

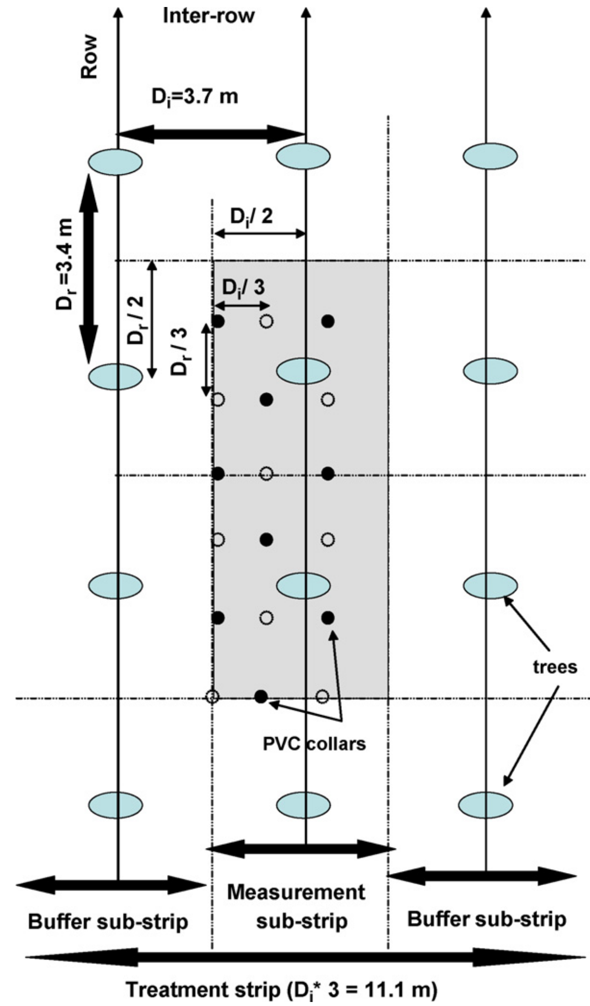

Fig. 2. Sampling grid scheme for soil respiration measurements indicating the 9 positions (filled circles) where the PVC collars were placed. There were 2 replicates of the 9 positions (18 PVC collars distributed around 4 trees) in the measurement sub-strip of each plot.

2.2. Above- and below-ground biomass and soil carbon contents of the savannah

In March 2005, before site preparation, above- and below-ground savannah biomass was measured in the three experimental blocks, in order to estimate the inputs of carbon to the soil from savannah residues during site preparation operations.

Table 1
Soil particle size distribution, soil bulk density, and carbon contents in superficial soil layers

Soil layers	0–5 cm	5–15 cm	15–25 cm	25–50 cm
Sand (%)	92.9 ± 2.1	92.0 ± 2.7	91.1 ± 2.8	89.5 ± 3.1
Silt (%)	4.9 ± 1.7	5.2 ± 2.1	6.1 ± 2.1	6.6 ± 2.0
Clay (%)	2.2 ± 0.5	2.8 ± 1.3	2.8 ± 1.2	3.9 ± 2.0
Bulk density	1.30 ± 0.08	1.37 ± 0.05	1.40 ± 0.06	1.43 ± 0.05
Carbon content (g C kg^{-1} soil)	6.91 ± 1.19	5.41 ± 1.15	4.33 ± 0.75	3.18 ± 0.63
Soil carbon density (kg C m^{-2})	0.45	0.74	0.61	1.13

Mean values and standard deviations are indicated (n = 24). Soil C concentrations were determined on finely ground (<200 μm) oven dried (40 °C) aliquotes of soils by dry combustion using an Elemental Analyzer CHN Fisons/Carlo Erba NA 2000 (Milan, Italy).

Aboveground biomass was estimated by clipping grasses on 36 locations (12 locations per block) within a 1 m^2 frame, and weighting them after a 72 h period at 65 °C. Root biomass in four soil layers (0–30, 30–60, 60–90 and 90–120 cm) was estimated at the same 36 locations, by soil coring. For each location, four soil cores (7 cm diameter and 30 cm long) were extracted in the superficial 0–30 cm, and two soil cores in each of the 3 deeper layers. Soil samples were sieved, and root fragments were washed, oven dried at 65 °C till constant weight, and weighed.

Initial savannah SOC contents and soil bulk densities were measured in August 2005 in the vicinity of the afforested area (Table 1). Soil sampling locations were distant by less than 50 m from our experimental area. SOC contents 1 year after afforestation were not assessed since detectable changes in SOC contents are not expected over such a short period of time because of soil spatial variability and further complications (such as changes in soil bulk density) that may result from the mechanical site preparation (Paul et al., 2002; Binkley et al., 2004; Smith, 2004).

2.3. Soil CO_2 efflux

Soil respiration (Rs) was measured over 1 year, at approximately 2-week intervals (23 measurement dates between April 1, 2005 and April 19, 2006) on PVC collars (10 cm in diameter and 7 cm in height), using the Li-8100 soil CO_2 flux system (LiCor Inc., Lincoln, NE, USA). We used a 10 cm (Li8100 102) respiration chamber, in which the increase in CO_2 concentration was recorded at 1 s intervals by the Li8100's infrared gas analyser. This chamber has a carefully designed pressure vent to prevent pressure gradients and wind incursion from outside the chamber or airflow out of the chamber (Venturi effect; Bain et al., 2005). A pneumatic system allows the chamber to lower above the soil collar and close automatically, thus minimizing mechanical disturbances during the measurements (LI-COR, 2005). The length of measurement was limited to 2 min, in order to avoid important chamber CO_2 concentration changes, and therefore to minimize potential soil CO_2 flux underestimation that may result from altered soil-atmosphere concentration gradients (Davidson et al., 2002). The set of equations used for flux computation was also designed to avoid such potential underestimation (LI-COR, 2005).

Eighteen PVC collars were installed in each plot (108 soil collars for the 6 plots), following a sampling grid scheme including nine positions in the inter-row (Fig. 2). The PVC collars were distributed around 4 trees in each plot (2 replicates of each position). Soil collars were installed at the end of March 2005, 3 days before the beginning of CO_2 efflux measurements. Soil collars were inserted 4–5 cm deep in the soil. This depth ensured collar stability, and was assumed to be sufficient to minimize potential soil respiration underestimation due to lateral diffusion of CO_2 (Hutchinson and Livingston, 2001; Davidson et al., 2002). Underestimation of soil CO_2 efflux due to root severing (Wang et al., 2005) did not occur since collars were installed before planting, i.e. before roots colonised the soil surface.

Volumetric soil water content of the 0–6 cm layer ($\theta_{0.06}$) was measured adjacent to the PVC-collars using a Theta Probe (Type ML2X, Delta-T Devices Ltd., Cambridge, UK). Measurements of soil water content and soil CO_2 efflux at the 108 locations were always performed during the same day.

2.4. Tree growth and above- and below-ground biomass in the plantation

Tree growth (tree height and diameter at 1.30 m when trees were higher than this threshold) was monitored from five stand inventories from October 2005 to May 2006. Below- and above-ground biomasses of trees were estimated from these stand inventories by means of allometric relationships between biomass of tree compartments and tree height and/or diameter as previously described (Epron et al., 2006). We used age-related equations that have been established for this *Eucalyptus* clone by Saint-André et al. (2005). We assumed that site preparation had no significant effect on the allometric relationships between tree size and the biomass of the various tree compartments, which is a reasonable assumption following Morris et al. (2006).

2.5. Data analyses

Savannah biomass in the three blocks before soil preparation was compared using one-way ANOVA (SAS). A two-way ANOVA was made for tree biomass at each inventory to test treatment and block effects. When the ANOVA indicated significant effects, the means were compared with Newman Keuls' multiple comparison tests. Soil respiration and soil water content were tested at each sampling date using the GLM procedure of the SAS software with the model:

$$V_{ijk} = \mu + \alpha_i + \beta_j + \gamma_k + \beta_j\gamma_k + \varepsilon_{ijk}, \quad (1)$$

where V_{ijk} is the studied variable (CO_2 efflux or soil water content) in the block i, the soil preparation treatment j, and the position in the inter-row k, μ is the general mean, α_i is the block i effect, β_j is the effect of the soil preparation j, γ_k is the effect of the position in the inter-row, $\beta_j\gamma_k$ is the interaction between the treatment and the position and ε_{ijk} is the residual value. Homogeneity of variances was tested at each sampling date by Levene's test and original values were log transformed when variances were unequal. The PDIFF statement of PROC GLM was used to compare the means when values were missing in the data set. Means are given with their standard deviations in the figures. The probability level used to determine significance was $P < 0.05$.

2.6. Soil carbon balance

Soil carbon balance during the first year after afforestation (ΔC) may be estimated as the difference between carbon inputs to the soil from savannah residues (S) and from litter production by the young eucalypt trees (L, resulting from leaf and fine root turnover), and soil carbon losses through heterotrophic soil respiration (i.e. soil CO_2 efflux resulting from the decomposition of savannah residues and soil organic matter), R_H.

Eighteen (6 per plot) 0.55 m^2 (0.74 m × 0.74 m) litter-traps were installed in each treatment for estimating above-ground litter production by young *Eucalyptus* trees, L_a. However, in both treatments, cumulated L_a during the first year of this experiment was very low (less than 5 g C m^{-2}), and was therefore neglected. Below-ground litter production has also been neglected. This assumption appears reasonable considering the low fine root biomass (especially the first 6 months), and a mean fine root life span of about 4–6 months (fine root turnover of about 2–3 at this age; Jourdan, personal communication). Therefore, ΔC was estimated as:

$$\Delta C = S_a + S_b - \int_0^{365} R_H(t)\,dt, \quad (2)$$

where S_a and S_b are the carbon inputs to the soil from above- and below-ground savannah residues.

S_a and S_b were estimated from measured above- and below-ground savannah biomass, assuming a carbon content of 0.5 g C g^{-1} DM. In the D treatment, above-ground savannah residues were incorporated into the soil during site preparation operations, whereas in the H treatment they were brought to the soil more progressively, through litter-fall after the plants were killed. However, this difference in timing was neglected since litter-fall and decomposition on the soil surface in the H treatment occurred rapidly, during the first weeks after planting. Therefore, even for the H treatment, CO_2 effluxes resulting from the decomposition of above-ground savannah residues were included in measured soil CO_2 effluxes (savannah residues that fell above soil respiration collars were manually placed inside the collars).

During the first weeks after treatment establishment, when root respiration by young eucalypts is likely to be negligible, heterotrophic soil respiration (R_H) can be estimated directly from measured soil CO_2 efflux ($R_H \approx R_s$). Over several months, however, the contribution of stand root respiration (R_A) to cumulated soil CO_2 effluxes may not be negligible, and R_H should therefore be assessed and deducted from R_s ($R_H = R_s - R_A$). Stand root respiration was estimated using specific root respiration rates measured between March and June 2005 on trees of the same age (7–12 months), at a nearby site planted with the same clone (Marsden, 2006):

$$R_A = [B_{fr}R_{fr30} + B_{mr}R_{mr30} + B_{cr}R_{cr30}]Q_{10}^{(T_s-30)/10}, \quad (3)$$

where R_A is the stand root respiration (μmol m^{-2} s^{-1}), B_{fr}, B_{mr} and B_{cr} are the fine (0–5 mm), medium (5–10 mm), and coarse root (diameter >10 mm) biomasses (g m^{-2}) estimated from stand inventories and age-related allometric equations established for this Eucalypt clone by Saint-André et al. (2005) (see above), and R_{fr30}, R_{mr30} and R_{cr30} are the mean specific root respiration rates (μmol g^{-1} s^{-1}) measured by Marsden (2006) and normalised at a reference temperature of 30 °C. The effect of soil temperature (T_s) on root respiration was estimated using the Q_{10} value (2.01) estimated by Marsden (2006).

Eq. (3) was applied at a semi-hourly time-step, using stand root biomass values (B_{fr}, B_{mr} and B_{cr}) that were linearly interpolated between each stand inventory. Since soil temperature was not monitored in these experimental plots, we used semi-hourly soil temperature (with thermocouples inserted at different depths between 2 and 30 cm) measured in 2005 and 2006 in a nearby (≈5 km) 3-year-old *Eucalyptus* plantation (at the 'Kissoko eddy-covariance' site; Marsden et al., in press). This extrapolation of soil temperatures from one stand to another should not have resulted in large errors: at a nearby site, Epron et al. (2006) found little difference (less than 0.9 °C on an annual basis) between soil temperature (0–10 cm) measured in a new eucalypt stand (0–1-year old) and an older (9–10-year old) adjacent eucalypt stand.

For each treatment, cumulated soil heterotrophic respiration was calculated as the difference between cumulated soil CO_2 efflux and cumulated root respiration. Cumulated soil CO_2 efflux was calculated using linear interpolations of R_s between each measurement date. At a nearby site where R_s was also measured at 2-week intervals, Marsden (2006) showed that annual estimates of cumulated soil CO_2 effluxes obtained using linear interpolations between measurement dates were less than 2.5% different from those calculated from R_s values interpolated using a strong relationship between R_s and soil water content, and daily measurements of soil water content.

3. Results

3.1. Savannah biomass

Total savannah biomass measured before site preparation was 856 g DM m^{-2}, without any significant difference between blocks. Above-and below-ground biomass, S_a, and S_b, were 231 and 625 g DM m^{-2} (mean root-shoot ratio of about 2.7), respectively. About 77% of the root biomass was localised in the superficial 0–30 cm soil layer (Fig. 3). This depth corresponds approximately to the depth subsequently disturbed by disk harrowing.

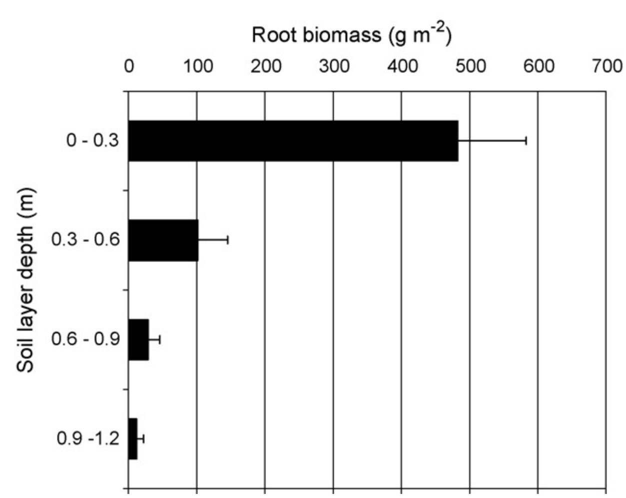

Fig. 3. Savannah root biomass distribution in the soil profile previous savannah destruction and tree planting.

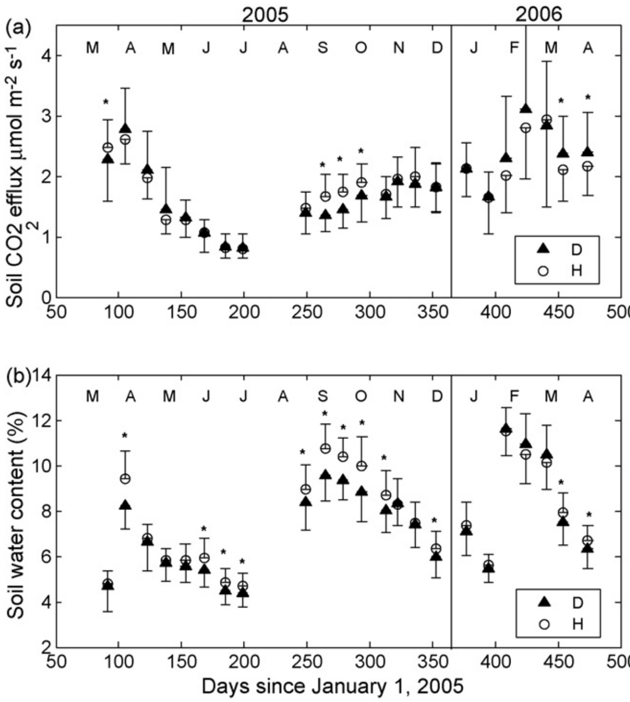

Fig. 4. Time course of (a) soil CO_2 efflux and (b) volumetric soil water content of the 0–6 cm soil layer ($\theta_{0.06}$), in D plots (soil disturbance treatment; filled black triangles) or H plots (herbicide treatment, open circles). Standard deviations ($n = 54$) are shown for both treatments (dotted lines for D treatment and continuous lines for H treatment). Stars indicate significant differences between treatments at the 5% threshold.

3.2. Soil water content and soil CO_2 efflux

The two site preparation treatments exhibited very similar temporal patterns of both soil CO_2 efflux and soil water content (Fig. 4a and b): the highest rates of soil CO_2 efflux (about 2.9 μmol m^{-2} s^{-1}) were observed during the rainy seasons of 2005 and 2006 when the soil water content of the 0–6 cm soil layer ($\theta_{0.06}$) was high, whereas the lowest rates (about 0.8 μmol m^{-2} s^{-1}) were observed during the dry season (June–September 2005). For most dates, there was no significant difference between soil CO_2 effluxes measured in D and H treatments. When slight significant differences ($P < 0.05$) were observed, soil CO_2 efflux was either higher in the H plots (at four occasions) or in the D plots (at two occasions). Cumulated soil CO_2 effluxes in H and D plots were not significantly different (Fig. 5).

Soil water content ($\theta_{0.06}$) ranged from 4.7 to 11.5% in H plots and from 4.4 to 11.6% in D plots. At several occasions, significantly higher values of $\theta_{0.06}$ were observed in H plots (Fig. 4b). The highest differences (about 1.2%) were observed when $\theta_{0.06}$ was high. The average difference between $\theta_{0.06}$ measured in the H and D plots was 0.36%.

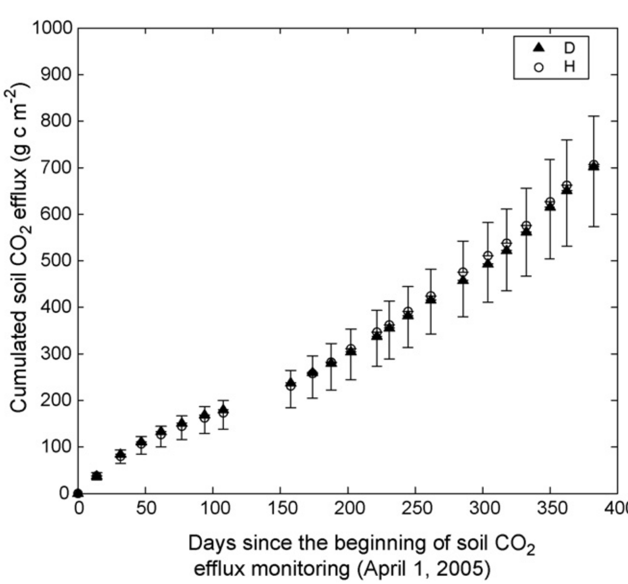

Fig. 5. Cumulated soil CO_2 efflux in the H and D plots. Standard deviations ($n = 54$) are shown for both treatments (dotted lines for D treatment and continuous lines for H treatment).

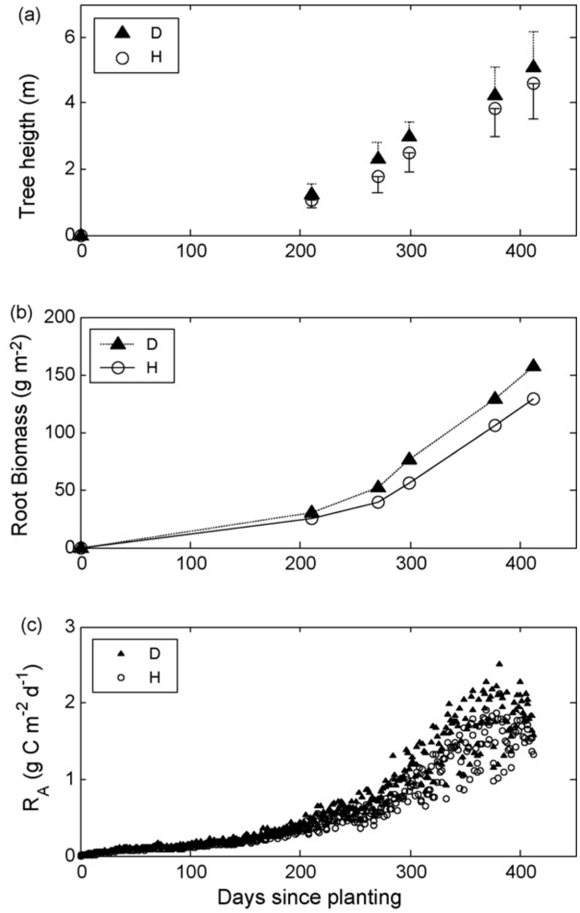

Fig. 6. Changes in tree height (a), root biomass (b) and root respiration (c) over 13 months after planting in H and D plots.

3.3. Tree growth

In contrast with soil CO_2 efflux, site preparation had a significant effect on tree growth (Fig. 6a). Tree growth was significantly enhanced by disk harrowing. Seven months after planting (at the end of October 2005), tree height was about 1.04 m in the H plots, and 1.25 m in the D plots. The difference in tree height measured in the two treatments increased from

0.21 m in October 2005 to 0.53 m in December 2005. This difference was then maintained or slightly decreased up to the age of 13 months (0.48 m in May 2006). When trees were 13 months old (in May 2006), mean tree total biomass (leaves + branches + stems + stump + roots) was 6.4 and 8.3 kg tree^{-1} in the H and D plots, respectively, and stand biomass was about 495 and 652 g DM m^{-2} (4.9 t ha^{-1} and 6.5 t ha^{-1}), for the H and D plots, respectively (mortality of 4 and 1% in the H and D treatments, respectively).

3.4. Soil carbon balance

Soil CO_2 efflux (R_s) was not increased by disk harrowing. At the end of the experiment (on April 19, 2006), cumulated R_s (over 384 days) in the H and D plots were 706 and 701 g C m^{-2}, respectively (Fig. 5). Even during the first months following site preparation, no detectable effect was observed: after 6 months, cumulated R_s reached 269 g C m^{-2} in both H and D plots (Table 2), and amounted to about 41% of annual soil respiration. Lower R_s during the first 6 months than during the six following months may be explained by lower soil water content (associated with the dry season from June to September 2005), and lower root respiration (Fig. 6c) due to low root biomass: 7 months after planting, estimated total root biomass was only 26 and 30 g DM m^{-2} in the H and D plots, respectively (Fig. 6b). Roots contributed to about 9% of soil CO_2 efflux during the first 6 months, and to about 27% of the annual soil CO_2 efflux. Cumulated R_A over 1 year was 25% higher in the D plots than in the H plots as a result of higher tree growth and root biomass.

Heterotrophic soil respiration (R_H) represented about 91%, and 73% of the soil CO_2 efflux cumulated over 6 months, and over 1 year, respectively, and remained stable from the first 6 months (245 g C m^{-2}) to the following 6 months (236 g C m^{-2}; Table 2). Cumulated R_H over 1 year in the D plots was 10% lower than in the H plots.

Table 2
Cumulated soil CO_2 efflux (R_s), root respiration (R_A), and heterotrophic soil respiration (R_H) during the first 6 months and the first year following treatment establishment

Treatment	0–6 months (g C m^{-2})	0–12 months (g C m^{-2})
R_s		
H	269	662
D	269	653
R_A		
H	22	158
D	26	197
R_H		
H	246	505
D	243	456
ΔC		
H	182	−77
D	185	−28

R_H was obtained as the difference between R_s and R_A. The soil C balance (ΔC) was estimated from Eq. (2).

Soil carbon inputs due to savannah residues were estimated at 428 g C m^{-2}, assuming a carbon content in savannah residues of 0.5 g C g^{-1} DM. This is less than the annual carbon outputs through heterotrophic respiration (505 and 456 g C m^{-2} in the H and D plots, respectively; Table 2). Therefore, the soil carbon budget during the first year after afforestation was slightly negative in both the H (−77 g C m^{-2}) and D (−28 g C m^{-2}) plots (Table 2).

4. Discussion

4.1. Effects of disk harrowing on early tree growth

The establishment (first planting) or re-establishment (replanting) phase of forest plantations commonly involves major mechanical disturbance of soil prior to planting using processes such as ploughing, disk harrowing and ripping (Turner and Lambert, 2000; Paul et al., 2002; Lincoln et al., 2007). The disturbance is undertaken because it is thought to enhance early tree growth through various mechanisms, including competition reduction, reduction of soil strength to allow better root penetration, improved water infiltration, and improvement in nutrient availability (Turner and Lambert, 2000). At our site, disk harrowing increased early tree growth by about 25% (total biomass). Similar increases in productivity after mechanical site preparation have often been reported (Fox, 2000; Morris et al., 2006; Jandl et al., 2007), and were explained by either improved soil physical conditions (porosity, lower soil penetration resistance allowing better root growth), lower competition (weed control), or improved nutrient and water availability (Collet et al., 1996; Morris et al., 2006; Lincoln et al., 2007). In our experiment, improved soil physical conditions were probably the main factor explaining the increase in tree growth, since competing vegetation was totally eliminated in both treatments (directed herbicide spraying throughout the year), and nutrient availability was probably not a limiting factor (trees were fertilised) nor a discriminative factor, since soil CO_2 efflux (and presumably nutrient release from residue and SOM decomposition) was not different between the two treatments (Fig. 5 and Table 2). At a nearby stand, Laclau et al. (2001) showed that 6.5 years after planting, the effect of subsoiling on soil strength and root distribution was still detectable. They also showed that soil strength increases sharply as the soil dries, reaching values >5 MPa during the dry season. In the H treatment, root growth may have been hindered by high soil strengths during the dry season, which started less than 3 months after planting, while the detrimental effect of soil mechanical resistance might have been alleviated by disc harrowing in the D treatment. In loblolly pine plantations, strong negative correlations between early tree growth and soil penetration resistance have been reported (Lincoln et al., 2007).

Disc harrowing may also have increased rainwater infiltration by decompacting the soil surface (Phiri et al., 2001; Diaz-Zorita et al., 2002), and by alleviating surface soil water hydrophobicity (Doerr et al., 1998). Severe surface soil hydrophobicity has been shown to exist in both savannah and *Eucalyptus* plantations (Laclau et al., 2001, 2004), and may partly result from the passage of fire, which occurs annually in these savannahs, and which is known to increase soil compaction (Snyman, 2003) and to produce hydrophobic substances that reduce soil infiltrability (Scott and Van Wyk, 1990; Snyman, 2003). Unfortunately, soil water content below 5 cm was not measured, so that the potential effect of disc harrowing on rainwater infiltration was not assessed.

Also, the long-term effect of disk harrowing on tree growth was not assessed in this study. In the same area, Nzila et al. (1997) found that soil sub-soiling before replanting increased early tree growth, but had no detectable effect 3 years after planting (unpublished data).

4.2. Effect of disk harrowing on soil CO_2 effluxes

The site preparation treatment had no effect on the seasonal pattern of soil CO_2 effluxes. The pronounced seasonal trend observed in both H and D plots, with average values during the dry season about two to three times lower than during the wet season, is similar to the trend observed in nearby *Eucalyptus* stands (Epron et al., 2004, 2006), or in tropical savannahs or pastures (Davidson et al., 2000; Chen et al., 2002), where a large part of the seasonal variations of total soil respiration (Davidson et al., 2000; Chen et al., 2002; Epron et al., 2004) and its heterotrophic component (Chen et al., 2003; Epron et al., 2006) was explained by the variations of soil water content. The annual soil CO_2 efflux (0.66 and 0.65 kg C m^{-2} $year^{-1}$ for the H and D treatments, respectively) was very similar to the heterotrophic soil CO_2 efflux estimated in a nearby *Eucalyptus* plantation (0.65 kg C m^{-2} $year^{-1}$; Epron et al., 2006), and similar to the heterotrophic soil CO_2 efflux estimated in a tropical savannah of northern Australia (0.72 kg C m^{-2} $year^{-1}$, Chen et al., 2003).

We hypothesized that soil disturbance by disk harrowing would increase soil CO_2 efflux through enhanced heterotrophic respiration, but the results did not support our hypothesis. These results contrast with the widespread belief that initial soil C loss following forest establishment could be reduced by the adoption of silvicultural practices that do not disturb the soil (Turner and Lambert, 2000; Zinn et al., 2002). They also contrast with the results of Mallik and Hu (1997) who found that soil CO_2 effluxes after the clearcut of a boreal mixed-wood were 28% higher in plots where the organic matter had been mixed with the mineral horizons, compared to plots with undisturbed soils, probably as a result of better soil aeration. Increases in soil CO_2 effluxes due to soil disturbance have also been reported for agricultural lands (e.g. Baker and Griffis, 2005; La Scala et al., 2005). A CO_2 flush is sometimes observed immediately after tillage, due to the transfer to the atmosphere of CO_2 stored in soil air space, as a result of the perturbation of the superficial soil structure, but this flush is generally short-lived (e.g. 10–24 h in Ellert and Janzen, 1999), resulting in insignificant effects on annual CO_2 effluxes (Roberts and Chan, 1990; Ellert and Janzen, 1999).

Soil disturbance is thought to increase soil respiration through an increase in residue and SOM decomposition because of a better soil aeration (Mallik and Hu, 1997). However, in these coarse-textured savannah soils (sand content >90%), soil aeration might not be a limiting factor. Disk harrowing may also increase the rate of organic matter decomposition by burying savannah residues into the soil, thus resulting in enhanced availability of fresh organic matter to soil micro-organisms (e.g. Thorburn et al., 2001), and possibly enhanced SOM decomposition as a result of the 'priming effect' (stimulation of the microbial activity through the addition of an easily available C source; Fu et al., 2000; Kuzyakov, 2006). However, aboveground savannah residues represented a modest input of organic matter to the soil, evaluated at 231 g DM m^{-2} ($\approx$115 g C m^{-2}), representing only 27% of total (above- and below-ground) savannah residues. Furthermore, the decomposition of aboveground savannah residues is rapid. In a nearby young *Eucalyptus* plantation established over a savannah in 2001, 76% of the aboveground residues and 85% of the root residues had decomposed after 1 year (Kazotti and Deleporte, unpublished results). Decomposition in 2005 was probably much faster than in 2001 due to wetter soil conditions (there was an exceptionally long dry season in 2001). Therefore, potential differences of decomposition rate between the two treatments would have had an insignificant effect on cumulative soil CO_2 effluxes.

Soil physical disturbance could potentially increase SOM mineralization and therefore soil respiration, by disrupting soil aggregates, thereby increasing the accessibility of intra-aggregates SOM to decomposers (Post and Kwon, 2000; Six et al., 2002). The effect of soil disturbance might therefore depend on both the importance of the macroaggregate-protected pool of OM, and the aggregate stability. In our study, the aggregate-protected pool of OM was not quantified, but this pool is known to be much lower in coarse textured soils than in fine textured soils (Feller and Beare, 1997). Feller and Beare (1997) reviewed several studies carried out on highly weathered sandy soils with a mineralogy dominated by low activity clay (LAC) minerals (i.e. 1:1 clay minerals such as kaolinites, as in this study) that found no difference in the quantities of mineralized C and nitrogen from intact 'aggregated' soils and the sum of that measured from dispersed particle-size fractions or from samples grinded into increasingly smaller particle-size fractions.

4.3. Soil carbon balance after afforestation

Several studies based on measurements (e.g. Turner and Lambert, 2000; Paul et al., 2002) and modelling (Paul et al., 2003) have reported an initial decline in soil C after afforestation of grasslands, mainly due to the cessation of litter inputs from the previous vegetation. This decrease is generally reversed after a few months or years, as the new planted forest develops and starts producing increasing amounts of litter (Paul et al., 2002, 2003).

The time required for recovery to previous soil C contents might depend on the magnitude of the initial decline. In the present study, the soil C balance after afforestation was assessed by comparing C inputs to the soil from savannah residues with

C losses through heterotrophic respiration (Eq. (2)). In their review on soil C changes following afforestation, Paul et al. (2002) found no significant effect of soil disturbance level, even in newly established (<10-year) plantations, and Baker et al. (2007) suggest that the effect of soil disturbance may sometimes have been over-estimated due to inadequate (shallow) soil samplings that do not account for the redistribution of carbon in the soil profile caused by tillage (e.g. soil-mixing). However, the importance of minimising soil disturbance to avoid unintended C losses is still stressed (Jandl et al., 2007). Here, we reported a net carbon loss during the first year after afforestation but the loss was not enhanced by soil disturbance. In contrast, we observed an unexpected slightly lower carbon loss in the D treatment that probably highlights uncertainties in estimated root biomass, i.e. in the autotrophic component correction.

The total C input to the soil (428 g C m^{-2}) from above-ground (115 g C m^{-2}) and below-ground (313 g C m^{-2}) savannah residues was outweighed during the 12 following months by the carbon loss through heterotrophic respiration (456–505 g C m^{-2}), leading to a slight decrease in soil C (−28 to −77 g C m^{-2}). The heterotrophic soil respiration (R_H) estimated at our site was lower than that reported by Chen et al. (2003) for a tropical savannah of northern Australia (720 g C m^{-2}). As discussed above, probably more than 76 and 85% of savannah above- and below-ground residues decomposed over a period of 1 year. This represented a CO_2 efflux of about 250–300 g C m^{-2} if we assume that 80% of the decomposed C was lost as CO_2 (microbial efficiency of 0.2). Therefore, about 60% of R_H would have resulted from the decomposition of savannah residues, and 40% (≈150–250 g C m^{-2}) from the decomposition of SOM.

The initial decrease in soil C (ΔC; Eq. (2)) during the first year after afforestation (−28 to −77 g C m^{-2} $year^{-1}$; Table 2) was low compared to the initial soil C loss reported in some studies (Turner and Lambert, 2000; Paul et al., 2002). Paul et al. (2003) found that the initial decline in soil C across a range of soil types and climates after afforestation on pastures was highly and positively correlated to the initial amount of soil C. For our site, their equation would predict an initial decrease in soil C of about −0.5% $year^{-1}$ (≈−11 g C m^{-2} $year^{-1}$) in the 0–30 cm soil layer, which represents a low C loss, consistent with our results. The low carbon contents measured at our site could be easily explained by the low (clay + silt) content, since strong correlations have often been reported between soil C and (clay + silt) contents (Feller and Beare, 1997; Mendham et al., 2002; Six et al., 2002; Zinn et al., 2005). But the annual occurrence of fire might also account for the low soil C content as already shown in a Zimbabwese savannah growing on coarse textured soils (Bird et al., 2000).

Soil carbon balance was not assessed after more than 1 year of age in this *Eucalyptus* stand. Results from other studies suggest that soil C may not increase (or may even decrease) after afforestation on very productive grasslands, but that increases in soil C can be obtained after afforestation on poorly productive grasslands, or degraded pastures (Paul et al., 2003; Lima et al., 2006). In Congo, measurements of soil respiration, litter production, soil C contents, $\delta^{13}C$ of SOM and of CO_2 respired by the soil are currently being made in a chronosequence of *Eucalyptus* stands, in order to assess the effect of savannah afforestation on soil C dynamics during the first *Eucalyptus* rotation.

5. Conclusions

At our site characterised by sandy LAC soils, a single soil disturbance with disk harrowing had no effect on soil CO_2 efflux and soil C balance after savannah conversion to *Eucalyptus* plantations, but significantly increased early tree growth, probably as a result of decreased soil penetration resistance allowing better root growth. We found a limited soil C loss during the first year after planting, which is consistent with the low initial soil carbon content in these annually burnt savannahs and which suggest a potential C sequestration for the long term. But other management practices such as fertilisation, rotation length, coppicing versus replanting, residue management and better fire prevention may have strong impacts on the C accumulation in the soil over successive rotations, and therefore warrant further research.

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