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Forest management effects on *in situ* and *ex situ* slash pine forest carbon balance

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ABSTRACT

In this study we analyzed the effect of silviculture on carbon (C) budgets in *Pinus elliottii* (slash pine) plantations on the southeastern U.S. Coastal Plain. We developed a hybrid model that integrates a widely used growth and yield model for slash pine with allometric and biometric equations determined for long-term C exchange studies to simulate *in situ* C pools. The model used current values of forest product conversion efficiencies and forest product decay rates to calculate *ex situ* C pool. The model was validated from a variety of sources, accurately simulating C estimates based on multiple measurement techniques and sites. Site productivity was the major factor driving C sequestration in slash pine stands. On high productivity sites, silvicultural schemes that promote sawtimber-type products are more suitable for increasing C storage (even not taking in account the consequent economical revenues associated with sawtimber production). When rotation length was increased from 22 to 35 years on unthinned and thinned stands, respectively, we estimated net increments of 26 and 20 MgC ha^{-1} in average C stock of the first five rotations. Even though *in situ* C pool in slash pine accounts for most of this net increment, C in sawtimber products increased from 8 and 14 to 23 and 24 MgC ha^{-1}, on unthinned and thinned stands, respectively. Thinning effects on net C stock depended on intensity and timing of intervention, mainly due to changes in diameter classes that promote higher proportion of long-lived products. Emissions associated with silvicultural activities, including transportation of logs to the mill, are small compared to the magnitude of net C sequestration, accounting for between 2.2 and 2.3% of gross C stock. This slash pine plantation C sequestration model, based on empirical and biological relationships, is appropriate for use in regional C stock assessments or for C credit verification.

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1. Introduction

Over the last three decades, greenhouse gas (GHG) emissions have increased by an average of 1.6% per year, with carbon dioxide (CO_2) emissions from the use of fossil fuels growing at an average rate of 1.9% per year (Nabuurs et al., 2007). Mitigation of atmospheric CO_2 requires an approach that combines CO_2 emission reductions with increasing CO_2 storage (Sundquist et al., 2008). Forests and forest management have the potential to play a crucial role in the mitigation of atmospheric CO_2 through increased carbon (C) storage. The fixation of atmospheric CO_2 into plant tissue is one of the most effective mechanisms for offsetting C emissions (Sedjo, 1989; Sedjo et al., 1997; Nabuurs et al., 2007). At a global scale, forests sequester 1.3–4.2 GtCO$_2$-eq year^{-1} and some global top-down models predict far larger mitigation potentials of up to 13.8 GtCO$_2$-eq year^{-1} in year 2030 (Nabuurs et al., 2007). In the United States, forests represent over 90% of the terrestrial C sink, which is equivalent to 12 to 16% of U.S. GHG emissions (U.S. EPA, 2005), and while this is substantial, forests have the potential to sequester at higher rates through the application of sustainable forest management approaches (Johnsen et al., 2001). Han et al. (2007) pointed out that the forests in southeast and south-central U.S. could potentially capture CO_2 equivalent to 23% of the regional GHG emissions. An important advantage of forest management approaches to CO_2 mitigation is that silvicultural technologies are well developed, in place, and inexpensive to apply, which would allow forest management to act as a short- to medium-term "bridge" during the development of other sequestration technologies and while society implements emission reduction programs (Fisher et al., 2007; Sundquist et al., 2008).

There are four primary ways that forest management can be used to mitigate atmospheric CO_2: (1) Increasing the amount of forestland through afforestation or reforestation; (2) Increasing time-averaged C density or carbon stock per unit land area; (3) Substituting forest products for more C intensive building materials or energy sources; and (4) Avoiding deforestation and forest degradation (Canadell and Raupach, 2008; U.S. EPA, 2005). Silvicultural approaches have the potential to greatly influence forest carbon stocks per unit area. In southern pines, modern site preparation, control of competing vegetation, fertilization and use of superior genotypes can increase productivity up to four-fold compared with productivity of mid 1950s plantations (Fox et al., 2007). Rotation

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length management is known to affect C stored in stand (Cooper, 1983; Cropper and Ewel, 1987) and the use of extended rotations has been proposed as an effective way to manage carbon budget of forests (Liski et al., 2001). For the purposes of C accounting, it is important to note that C stocks in managed forest can be divided in two major pools: *in situ* C in standing biomass and soil organic matter and *ex situ* C sequestered in products created from harvested wood (Marland and Marland, 1992; Johnsen et al., 2001). Both *in situ* and *ex situ* pool size can be influenced by forest management.

In this study we analyzed the effect of silviculture on C budgets in slash pine plantations on the southeastern U.S. Coastal Plain. We developed a hybrid model for the analysis that integrates a growth and yield model for slash pine (Pienaar et al., 1996; Yin et al., 1998) with allometric and biometric equations determined for long-term C exchange studies (Gholz et al., 1986; Clark et al., 2004; Bracho et al., in preparation) to simulate *in situ* C pools. The model used current values of forest product conversion efficiencies and forest product decay rates to calculate *ex situ* C pool dynamics. Because silvicultural operations emit CO_2 (Markewitz, 2006), the model also simulated the C costs of the various tested scenarios. We hypothesized that (1) lengthened rotations would increase time-averaged C stocks; (2) regimes incorporating longer rotations and thinning would maximize accumulation of C in *ex situ* product pools; and (3) the C cost of silvicultural operations would be far outweighed by increased C sequestration resulting from the operations.

2. Materials and methods

2.1. Models

Growth and yield models were combined with allometric and biometric equations to estimate C fluxes and stocks for slash pine (*Pinus elliottii* var *elliottii* Engelm.) plantations in the SE United States. We used slash pine growth and yield models reported by Pienaar et al. (1996) and Yin et al. (1998), modified to allow the use of multiple fertilizations (Martin et al., 1997) and thinning (Bailey et al., 1980; Pienaar, 1995). These models predict stand growth in basal area (BA, $m^2\,ha^{-1}$), total volume (V, $m^3\,ha^{-1}$), dominant height (H, m), quadratic mean diameter (QMD, cm) and mortality, using as inputs soil type from the Cooperative Research In Forest Fertilization (CRIFF) classification (Kushla and Fisher, 1980), site index (SI, m), number of trees per ha at planting (N, trees ha^{-1}), type of site preparation (bedding and herbicide treatments), fertilization age and thinning timing and intensity (as a removal percentage from the surviving trees before thinning). Table 1 presents a list of functions used for growth and yield modeling.

At each age, allometric equations (Table 2; equations derived from pooled datasets of *R*. Gorman, personal communication and Garbett, 1977) were used to estimate aboveground stand biomass from *QMD* and number of surviving trees simulated by the growth and yield model. Coarse root biomass was estimated from equation reported by Santantonio et al. (1977). Several studies support the use of that allometric relationship to estimate coarse root on slash pine plantations (Gholz and Fisher, 1982; Gholz et al., 1985a; Clark et al., 1999; Shan et al., 2001; Clark et al., 2004). Fine root biomass was estimated from linear interpolation of values reported by Gholz et al. (1986). The estimation of stand biomass from *QMD* using non-linear, single-tree allometric equations is biased (Duursma and Robinson, 2003), so we used inventory data from years 1 to 17 (Powell et al., 2008) to calculate correction factors (CF, Table 2) for each biomass component as a function of stand age.

Slash pine leaf area index (LAI, $m^2\,m^{-2}$) was estimated from a 17-year chronosequence of litterfall (Bracho et al., in preparation) and fitted to stand *BA* measured yearly on permanent plots in the same site (Powell et al., 2008) by using a logistic model ($R^2 = 0.91$; Table 2). Understory biomass accumulation (B_U; Mg ha^{-1}) and litterfall biomass accumulation (B_L; Mg ha^{-1}) were estimated from

Table 1
Equations used for growth and yield modeling (English units).

Parameter	Equation	Source
H	$= 1.3679 \times SI \times (1 - e^{-0.07345\times A})^{1.804} + (0.678Z_1) + 0.546Z_2) + 1.395Z_2 - 0.412Z_1Z_3) \times A \times e^{-0.0691\times A}$	(4)
$R_H(A)$	$= (0.375379 \times T_1 + 0.652099 \times T_2) \times (A - A_f) \times e^{-(A-A_f)\times(0.154525\times T_1+0.100104\times T_2)}$	(3)
N_2	$N_1 \times e^{(-0.0041-0.0019\times Z_t)\times(A_2^{1.345}-A_1^{1.345})}$	(1)
BA	$= e^{-3.394-\frac{35.668}{A}} \times H^{1.336+\frac{6.205}{A}} \times N^{0.366+\frac{3.155}{A}} + (0.557Z_B + 0.436Z_1 + 2.134Z_3 - 0.345Z_1, Z_2) \times A \times e^{-0.09\times A}$	(4)
$R_{BA}(A)$	$= (1.12915 \times T_1 + 1.68711 \times T_2) \times (A - A_f) \times e^{-(A-A_f)\times(0.09793\times T_1+0.073688\times T_2+0.046995\times S_1+0.055385\times S_2)}$	(3)
BA_t	$BA_b \times \left(\frac{N_t}{N_b}\right)^{1.268}$	(1)
CI	$= 1 - \frac{BA_{at}}{BA_U}$	(1)
CI_2	$= CI_1 \times e^{-0.093\times(A_2-A_1)}$	(1)
BA_{t_2}	$= BA_{U_2} \times (1 - CI_2)$	(1)
V	$= H^{0.82} \times N^{-0.017-\frac{0.32}{A}} \times BA^{1.016+\frac{0.501}{A}}$	(4)
$V_{d,t}$	$= V \times e^{-0.52\left(\frac{t}{QMD}\right)^{3.84} - 0.69\times N^{-0.12\left(\frac{d}{QMD}\right)^{5.72}}}$	(2)
In (D_{24})	$= 40.32 - 0.4244 \times \ln(H) + 0.0725\ \ln(N) + 0.7756\ \ln(B/N)$	(2)
D_{24a}	$= -0.6617 + (0.2026 \times D_{24b}) + (0.8064 \times QMD_b) + (1.3855 \times BA_{tp})$	(5)

Note: H is dominant height (ft); A is stand age (years); SI is site index (ft); $Z_1 = 1$ if fertilized at planting, 0 otherwise; $Z_2 = 1$ if bedded (site preparation), 0 otherwise; $Z_3 = 1$ if herbicide was used, 0 otherwise; $R_H(A_i)$ is the dominant height response to fertilization (ft above unfertilized); $T_1 = 1$ if fertilization treatment was N only, 0 otherwise; $T_2 = 1$ if fertilization treatment was N and P, 0 otherwise; A_f is age at fertilization (years); N_2 is surviving trees per acre at age A_2; N_1 is surviving trees per acre at age A_1; $Z_t = 0$ if not thinned, =1 if thinned at or prior at age A_1; BA is basal area ($ft^2\ acre^{-1}$); $Z_B = 1$ if burned, 0 otherwise; $R_{BA}(A)$ is basal area response to fertilization ($ft^2\ acre^{-1}$ above unfertilized); $S_1 = 1$ if in soil group C, 0 otherwise; $S_2 = 1$ if in soil group D, 0 otherwise; BA_t is basal area removed at thinning ($ft^2\ acre^{-1}$); BA_b is basal area before thinning ($ft^2\ acre^{-1}$); N_t is number of trees per acre removed at thinning; N_b is number of trees per acre before thinning; CI is competition index at thinning age; BA_{at} is basal area after thinning ($ft^2\ acre^{-1}$); BA_U is basal area in the unthinned counterpart ($ft^2\ acre^{-1}$); CI_2 is competition index at age A_2; CI_1 is competition index at age A_1; BA_{t2} is the basal area in the thinned stand at age A_2 ($ft^2\ acre^{-1}$);; BA_{U2} is the basal area in the unthinned counterpart at age A_2 ($ft^2\ acre^{-1}$); V is total outside bark stem volume ($ft^3\ acre^{-1}$); $V_{d,t}$ is merchantable volume ($ft^3\ acre^{-1}$) of trees with DBH ≥ d inches to a merchantable diameter t inches outside bark; QMD is quadratic mean diameter (inch); D_{24} is the 24th percentile of the diameter distribution of an unthinned plantation (inch); D_{24a} is the 24th percentile of the diameter distribution after thinning (inch); D_{24b} is the 24th percentile of the diameter distribution before thinning (inch); QMD is quadratic mean diameter before thinning (inch); BA_{tp} is the basal area removed at thinning as a proportion of basal area before thinning. Sources: (1): Pienaar (1995); (2): Pienaar et al. (1996); (3): Martin et al. (1997); (4): Yin et al. (1998); (5): Bailey et al. (1980).

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Table 2
Allometric and biometric equations.

Component	Equation
B_C	$\ln(B_C) = -3.4466058 + 2.2321092 \times \ln(DBH)$
B_{AG}	$\ln(B_{AG}) = -2.5563158 + 2.5209397 \times \ln(DBH)$
B_{CR}	$\log_{10} B_{CR} = -1.8899 + \log_{10}(DBH)$
B_{FR}	If Age < 9: $B_{FR} = 0.3056 \times A$
	If Age ≥ 9: $B_{FR} = 0.435 + 0.2572 \times A$
CF B_C	$\mathrm{CF}\,B_C = \frac{0.988385}{1-0.09205 \times e^{-0.584309 \times A}}$
CF B_{AG}	$\mathrm{CF}\,B_{AG} = \frac{1.022265}{1-2.50759 \times e^{-1.310504 \times A}}$
CF B_{BG}	$\mathrm{CF}\,B_{BG} = \frac{0.959876}{1-0.04604 \times e^{-0.537546 \times A}}$
LAI	$\mathrm{LAI} = \frac{6.2683942}{1+23.06668 \times e^{(-0.40444316 \times \mathrm{BA})}}$
$B_{U(LAI)}$	$B_{U(LAI)} = \frac{0.88242377+14.925223}{1+1.5801704 \times \mathrm{LAI}+0.8650196 \times \mathrm{LAI}^2}$
$B_{L(LAI)}$	$B_{L(LAI)} = 0.78628459 \times \mathrm{LAI}^{1.0516138}$
SG	$SG = 0.329 \times A^{0.1557}$

Note: B_C is crown biomass (kg tree^{-1}); B_{AG} is aboveground biomass (kg tree^{-1}); B_{CR} is coarse root biomass (kg tree^{-1}); from Santantonio et al. (1977); B_{FR} is fine root biomass (Mg ha^{-1}); CF B_C is correction factor for canopy biomass estimation bias: CF B_{AG} is correction factor for aboveground biomass estimation bias (stem + crown); CF B_C is correction factor for below ground biomass estimation bias (fine + coarse roots): LAI is Leaf Area Index (m^2 m^{-2}); B_U is understory biomass (Mg ha^{-1}); B_L is litterfall (Mg ha^{-1}); *SG* is specific gravity; *DBH* is diameter at breast height (cm); *A* is stand age (years).

17-year biomass chronosequence measurements (Bracho et al., in preparation). Finally, LAI-dependent annuals B_U and B_L models were obtained using non-linear model fitting ($R^2 = 0.90$ and 0.71, respectively; Table 2). A decay rate of 15%/year mass loss was assumed (Gholz et al., 1985b; Gholz et al., 1986; Gholz et al., 1991).

Standing dead trees estimated from mortality equations (Pienaar, 1995; Yin et al., 1998) were incorporated into the dead component of total biomass. A large fraction of the stand mortality occurs in diameter classes below the median, due to the effects of resource competition on suppressed and weak trees (Martin and Jokela, 2004). Using diameter distribution models reported by Pienaar et al. (1996) for unthinned and thinned plantations, diameter at percentile 24th (D_{24}, cm) was determined for each age (Table 1). This diameter class was assumed to better represent the observed diameter class of dying trees (Martin and Jokela, 2004). Biomass of dying trees was computed in the same way as standing biomass, but D_{24} at the previous year was used instead of *QMD* in order to estimate individual tree biomass.

The effect of thinning on C fluxes of forest floor and understory biomass was also incorporated into the model. At the time of thinning, reductions in slash pine LAI were set to be proportional to reductions in *BA* due to thinning and therefore forest floor and understory biomass were affected due to their LAI-dependence (Table 2). Logging slash from thinned trees was also incorporated into flux calculations; thinning slash (root and crown biomass of extracted trees) was determined at each thinning and incorporated into the dead biomass pool.

Carbon mass (MgC ha^{-1}) was calculated by using an average C content of 50% for slash pine and understory biomass components (Clark et al., 1999; Smith et al., 2006).

2.2. Model validation

Validation of model results was carried out by comparing model outputs against published data of (i) aboveground C accumulation in live slash pine biomass (MgC ha^{-1}), (ii) forest floor and total C accumulation (not including soil) (MgC ha^{-1}) and (iii) net ecosystem production (NEP; MgC ha^{-1} year^{-1}).

The aboveground live slash pine C data was obtained from an extended dataset that includes estimates from young (from age 2) to mature (up to age 34) stands. The data was extracted from: (i) a chronosequence study (from age 2 to 34) reported by Gholz and Fisher (1982); (ii) 7-, 11- and 17-year-old stands reported by Gholz et al. (1986), Shan et al. (2001) and Clark et al. (2004); and (iii) 3 selected ages (5, 14 and 23 years) from a long-term slash pine productivity study (Jokela and Martin, 2000; Vogel et al., in preparation). Data from Shan et al. (2001), Jokela and Martin (2000) and Vogel et al. (in preparation) includes plots with and without weed control. In the case of the 34-year-old plots described by Gholz and Fisher (1982) we used the average of the 3 plots reported in order to avoid pseudo replication problems.

Forest floor and total C (not including soil C) estimates of slash pine stands of 7-, 11-, 17-, 25- and 27-year-old as well as after clear-cut estimates were obtained from Gholz et al. (1986), Shan et al. (2001) and Clark et al. (2004). Total C includes above and below ground estimates of live slash pine trees plus understory biomass plus forest floor (including litter and dead trees).

The model was also validated against net ecosystem production data from eddy-covariance measurements ($\mathrm{NEP_{EC}}$) from Bracho et al. (in preparation). The C flux studies have been carried out for more than 14 years in two sites in North Central Florida, in two commercial slash pine plantations, managed for pulpwood production, located approximately 15 km northeast of Gainesville, Alachua County, Florida, USA (29°44′ N, 82°9′30″ W) (Clark et al., 1999, 2004; Gholz and Clark, 2002; Bracho et al., in preparation). The first research site was established in a 24-year-old slash pine plantation in 1996. The site was harvested in 1998, double-bedded, treated with herbicide and replanted at approximately 1864 trees ha^{-1} in December 1998–January 1999, and fertilized during fall 2002. The second research site was established in an 8-year-old slash pine plantation in 1998. The site was replanted early in 1990 at approximately 2196 trees ha^{-1}, following clear cutting in 1988–1989, and fertilized in August, and December 2001 (Clark et al., 1999, 2004). Reported values of C stocking in the same site (Clark et al., 2004) were also compared with outputs from model. Forest floor and coarse wood debris biomass after harvest are those reported by Clark et al. (2004). Based on this study, the model decay rate was increased to 20% for first year after clear cutting and maintained constant at 15% (Gholz et al., 1985b; Gholz et al., 1986; Gholz et al., 1991) for the rest of the rotation.

Initial parameters of the model, such as *N*, SI and silvicultural management options (soil preparation, weed control, fertilization and thinning) were set equal to those values reported in each reference cited for each comparison.

2.3. Ex situ wood product pools

Harvested roundwood (from thinnings or clear-cuts) was assigned to three main product classes depending on stem DBH and merchantable diameter (Table 3); sawtimber (ST), chip-and-saw (CNS) and pulpwood (PW) using the model proposed by Pienaar et al. (1996) and Yin et al. (1998) (Table 1). Harvest efficiency of 87% of *V* was assumed (Bentley and Johnson, 2004). Merchantable volume inside bark was calculated for each stand age and product volume was transformed to biomass (Mg ha^{-1}) by multiplying by average specific gravity (*SG*) at a given age. An exponential model that correlates *SG* and age (Larson et al., 2001) was fit using published values of whole core weighted *SG* and age (Gibson et al., 1986; Harding, 2008; Howard, 1972; Lowerts and Zoerb, 1989; Miller, 1959; Sohn and Goddard, 1974). The model explains 64% of variability in *SG* with stand age (Table 2). A C content of 50% was also used to calculate C mass of each product type.

Industrial conversion efficiencies of 65, 65 and 58% were assigned to ST, CNS and PW, respectively (Table 3) following Peter et al. (2007), Spelter and Alderman (2005) and Smith et al. (2006). All the product types were divided into four lifespan categories

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Table 3
Wood product characteristics.

Product	Product proportion by lifespan category (%)				Conversion efficiency (%)	Product class diameter (cm)	
	Long	Medium-long	Medium-short	Short		*d*	*t*
	(50)	(16)	(4)	(1)			
ST	50	25	0	25	65	28	15
CNS	25	25	0	50	65	20	15
PW	0	0	33	67	58	10	5

Note: ST: Sawtimber; CNS: Chip and Saw; PW: Pulpwood; *d*: minimum DBH; *t*: Merchantable diameter. Values in parenthesis indicate average half-life for class (years).

(Table 3) according to the classification proposed by Liski et al. (2001) and Gundimeda (2001) and adapted to slash pine utilization patterns in the SE United States (Birdsey, 1996; Harmon et al., 1996; Row and Phelps, 1991, 1996).

2.4. Silvicultural management scenarios

To analyze the effect of silvicultural management and rotation length on C sequestration, we simulated C flux under four different scenarios for standard conditions of slash pine plantations established in spodosoils in North Florida (E. Jokela, personal communication). Initial parameters used were: SI = 22 m, N = 1500 trees ha^{-1}, bedding and weed control at planting and 178 kg ha^{-1} diammonium phosphate fertilization at age 4 years. In addition to this initial NP fertilization, midrotation fertilization at age 14 years was also included in the model for sawtimber production scenarios (140 kg ha^{-1} diammonium phosphate + 430 kg ha^{-1} urea). Initial C accumulated in forest floor and coarse wood debris from previous rotation was assumed to be 46.66 MgC ha^{-1} (Clark et al., 2004). Thinning intensity removal was set as 33% of the living trees.

The four scenarios selected were:

i) Pulpwood production (PP). No thinning; clear-cut harvest at age 22 years.
ii) Sawtimber production short rotation (ST1). One commercial thinning at age 14 years and final harvest at age 22 years.
iii) Sawtimber production long rotation (ST2). One commercial thinning at age 14 years and final harvest at age 35 years.
iv) Sawtimber production long rotation (ST3). Two commercial thinnings (at age 14 and 22) and final harvest at age 35 years.

2.5. Carbon emissions of transportation and silvicultural activities

Carbon emitted in transportation of raw material from the forest to the mill (MgC emitted from fossil fuels per MgC in raw material transported) was estimated according to White et al. (2005), assuming an average distance of 100 km from forest to mill, load per logging truck of 24 m^3 and fuel economy of diesel logging truck of 2.6 km l^{-1} (Table 4). Carbon emitted by silvicultural activities (MgC ha^{-1}) was determined from Markewitz, 2006 (Table 4). The Markewitz (2006) C emission estimates include fuel and lubricant consumption of machinery and emissions associated with manufacture of fertilizer and herbicide.

2.6. Sensitivity analysis

A sensitivity analysis was carried out to determine the effects of changes in key parameters on total C balance. The effect of site quality was assessed by evaluating the model under contrasting SI of 15 and 30 m, which corresponds to the full range of site quality observed in slash pine plantations in SE United States (E. Jokela, personal communication). Initial stand density effect was evaluated by running the model under contrasting initial planting density of 750 and 2250 trees ha^{-1}. Decomposition rate effect was evaluated by using decay rates of 10 and 20%, values beyond the extremes from rates of 12 and 17% reported by Gholz et al. (1985b), Gholz et al. (1986) and Gholz et al. (1991). Sensitivity to industrial conversion efficiency was assessed by increasing or decreasing the default values by 5% (Table 3). Average product lifespan effect was evaluated by changing the proportion of products in different lifespan classes. In the case of ST and CNS, the proportion of product in the long life class was changed by 25% step up and down and distributing the residual proportion in equal parts on the rest of lifespan classes. Rotation length effect, besides silvicultural management scenarios tested, was assessed by evaluating the model under PP scenario for 18 and 35 years rotation lengths.

For all simulations, estimations of C stock are reported as the average of all yearly values from the first five rotations.

3. Results

3.1. Model validation

There was good agreement between modeled (NEP$_{EST}$) and measured with eddy-covariance technique (NEP$_{EC}$) net ecosystem production (Fig. 1a). There was no statistical difference between average NEP$_{EST}$ and measured NEP$_{EC}$ from year 1 to 17 (P = 0.823), averaging 3.70 and 3.61 for NEP$_{EST}$ and NEP$_{EC}$, respectively. Between ages 6 and 17 there was also no statistical difference in NEP (P = 0.576), averaging 5.77 and 5.46 MgC ha^{-1} year^{-1} for NEP$_{EST}$ and NEP$_{EC}$, respectively.

Estimations of C stock in aboveground live pine were also well correlated with reported values for stands different in age and productivity (R^2 = 0.98; Fig. 1b). The intercept of that relationship was not different to 0 (P = 0.38), but the slope of 0.952 was slightly different to 1 (P = 0.047). In this analysis we did not include the three circled outlier points from Gholz and Fisher (1982) in the valida-

Table 4
Carbon use in silvicultural activities and product transport to mills.

Activity	Description	C use (MgC ha^{-1})
Site preparation	Raking or spot piling + Weed control (application + product) + Bedding	0.095
Planting	Machine planting	0.101
Initial fertilization (age 4)	178 kg ha^{-1} diammonium phosphate	0.075
Mid-rotation fertilization (age 14 and up)	107 kg ha^{-1} diammonium phosphate + 343 kg ha^{-1} urea	0.268
Thinning	Commercial thinning	0.156
Final harvest	Clear cutting at rotation age	0.156
Transport[a]	Average for 24 m^3 load capacity	0.0026

[a] Carbon use for transport is expressed in MgC used per m^3 transported.

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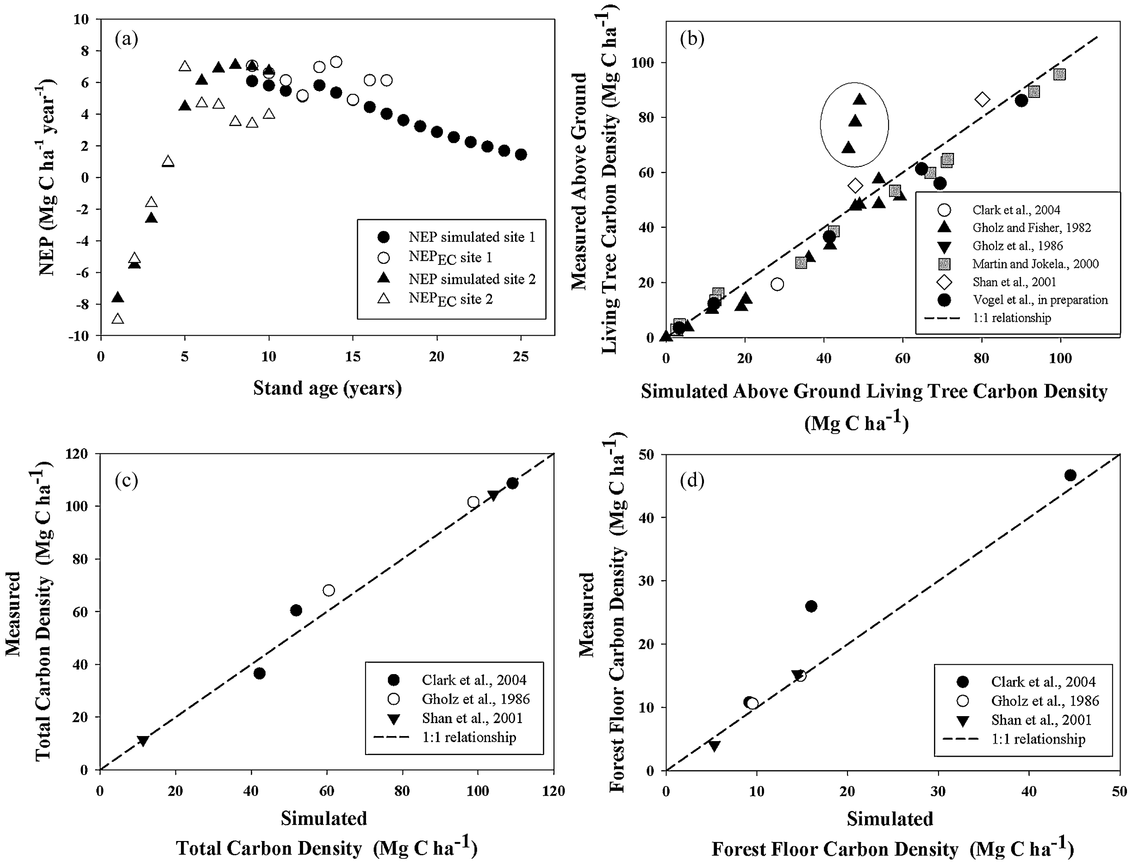

Fig. 1. Validation of model outputs: (a) Time series of simulated versus observed NEP at 2 sites; (b) Simulated versus measured aboveground living tree C stock (circled values were not included in validation regressions, see text for details); (c) Simulated versus measured total ecosystem C stock (not including soil); (d) Simulated versus measured C stock in forest floor.

tion regression (Fig. 1b). Those plots (2 of the 26-year-old plots and one of the 34-year-old plots) are recorded as having dominant heights of 16.7, 16.3 and 16.7 m, respectively (Gholz and Fisher, 1982), resulting in an effective site index of 16.3, 15.9 and 14.2 m. These very low site indices are not consistent with the high biomass values recorded for these plots. Unfortunately, it is impossible to determine the likely sources of error in the measurements collected more than 20 years ago.

Simulated total ecosystem C stock (not including soil C) was in good agreement with reported values (Fig. 1c). The intercept and slope of that relationship were not statistically different from 1 and 0, respectively ($P = 0.73$ and 0.95, respectively). For the same sites, forest floor C stock model estimates (Fig. 1d) were also in good agreement with measured values. The intercept and slope of that relationship were also not statistically different from 1 and 0, respectively ($P = 0.84$ and 0.41, respectively).

3.2. Silvicultural management effect on C sequestration

Under conditions used for simulations, net C stock, which corresponds to total C *in situ* (slash pine + understory + forest floor + standing dead trees) plus total C *ex situ* (C in woody products ST + CNS + PW) minus total C cost of silvicultural activities (including transport), for first 5 rotations averaged 105, 101, 122 and 126 MgC ha^{-1} for PP, ST1, ST2 and ST3, respectively (Fig. 2). *In situ* C stock accounted for 72, 69, 70 and 64% of gross C sequestration (not including silvicultural C cost), for the same silvicultural scenarios, respectively. Across silvicultural scenarios, forest floor averaged 23 MgC ha^{-1} and understory averaged 2.4 MgC ha^{-1}. The magnitude of emissions associated with silvicultural activities (including transport) was between 2.1 and 2.3% of gross C stock. The relative impact on C sequestration of different woody products depends on silvicultural scenario; ST accounted for 28, 44, 62 and 72% of total C *ex situ* for PP, ST1, ST2 and ST3, respectively, while CNS followed an opposite trend accounting for 63, 47, 33 and 24% for the same silvicultural scenarios, respectively.

For 22-year rotations, thinning 33% of living trees at age 14 decreases net C stock by 3.8 MgC ha^{-1}. This 3.6% decrease is explained principally by 5.2 MgC ha^{-1} reductions in slash pine C sequestration that counteracts the increase of 2.1 MgC ha^{-1} in *ex situ* C. Extending the rotation length to age 35 years with one thinning at age 14 (ST2) increase average net C stock by 17.5 and 21.3 MgC ha^{-1}, if compared with PP or ST1, respectively. If

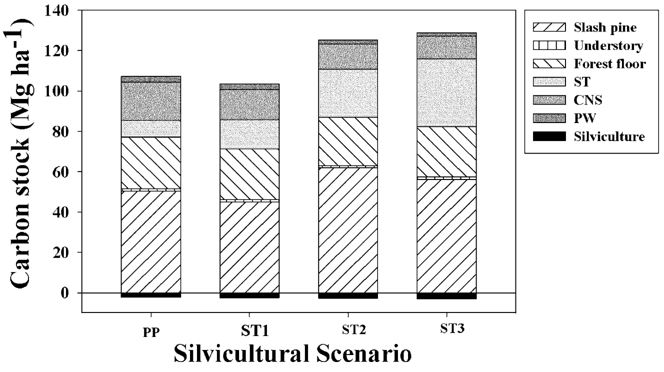

Fig. 2. Average carbon stock for slash pine plantations for first 5 rotations under different silvicultural scenarios.

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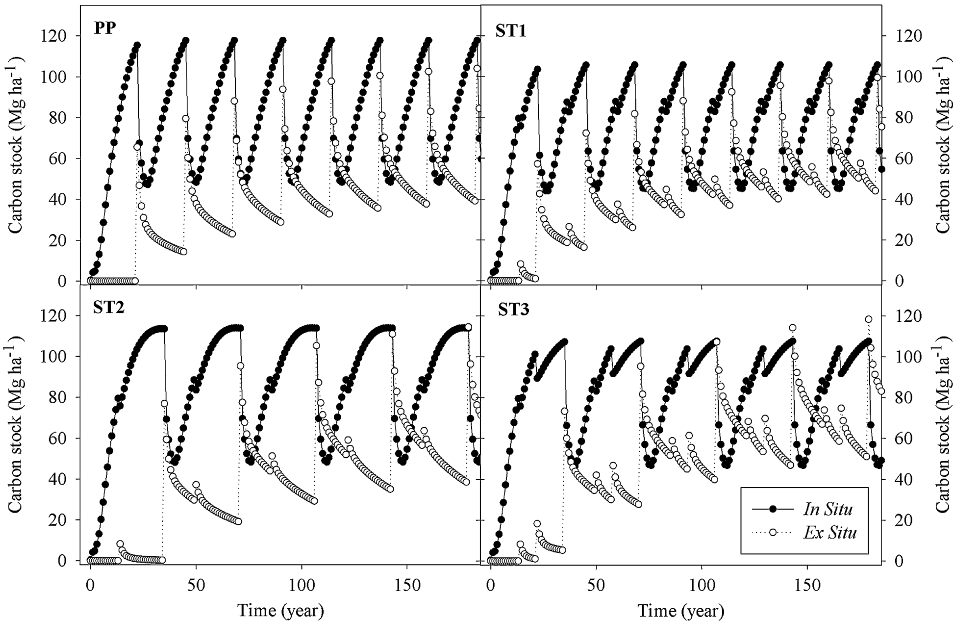

Fig. 3. Carbon stock for slash pine plantations under different silvicultural scenarios for a 180-year simulation period.

compared with PP, this larger C stock in ST2 is caused by larger accumulation in slash pine (11.6 MgC ha^{-1} increment) and woody products (8.0 MgC ha^{-1} net increment) C stock. When compared with ST1 the net effect on slash pine C sequestration is even larger, averaging 16.8 MgC ha^{-1} more C, and to a lesser extent on *ex situ* C (5.9 MgC ha^{-1} net increment). If a second thinning is carried out at age 22 on the extended rotation scenario (ST3), net C stock increases by 21.1 and 3.5 MgC ha^{-1}, if compared with PP or ST2, respectively. Most of this increase is due to larger C accumulation in woody products rather than *in situ* C stock. Net C sequestration in ST products increases 25.3 and 9.9 MgC ha^{-1} and C in CNS is reduced by 7.6 and 1.4 MgC ha^{-1}, when compared with PP and ST2, respectively. *In situ* C accumulation is increased by 5.1 MgC ha^{-1} or reduced by 4.8 MgC ha^{-1} if compared with PP and ST2, respectively. Woody products transport accounts for 1.06, 1.06, 1.28 and 1.42 MgC ha^{-1} for PP, ST1, ST2 and ST3, respectively, representing about 46 to 48% of total silvicultural C emissions.

At rotation age, *in situ* C stock was 115.4, 103.5, 113.4 and 107.3 MgC ha^{-1} for PP, ST1, ST2 and ST3, respectively. Total woody products C stock increases each rotation from 65.6, 57.2, 76.8 and 73.3 MgC ha^{-1} at first rotation, up to 97.7, 92.4, 114.4 and 118.3 MgC ha^{-1} at the end of 5th rotation, for the same silvicultural scenarios, respectively (Fig. 3). In general, after 200–250 years, C flux in woody products converges to stable values, reaching quasi-equilibrium maximum and minimum values of 42.6 and 107.4, 47.9 and 103.3, 43.3 and 119.2 and 57.0 and 124.1 MgC ha^{-1} for PP, ST1, ST2 and ST3, respectively. This implies that in the long term, for extended rotations, maximum and minimum C storage fluxes are larger in woody products rather than *in situ* (Fig. 3).

Differences in tree size (diameter and height) and number of trees remaining due to different thinning and rotation age scenarios create different woody product pools with different lifespans (Table 3). While PW represents 9.2, 8.6, 4.5 and 3.7% of total C extracted in products at harvest, ST accounts for 28.1, 44.4, 62.4 and 72.2% of that C on PP, ST1, ST2 and ST3, respectively. In general, C stored in products derived from PW (paper, packing material, office supplies, etc.) have a negligible effect on net C sequestration; between harvest events (thinning or clear cutting) the amount of C stored diminished towards 0 MgC ha^{-1}, while C stored in CNS and ST increased between harvests (Fig. 4).

3.3. Sensitivity analysis

With all other parameters held constant, site quality (or potential productivity) reflected in SI of the stand, was the major control of C sequestration (Table 5). On low productivity sites (e.g. SI = 15 m) net C stock is between 40.5 and 43.9% lower than in default site quality (SI = 22 m), on the other hand, in high productivity sites (e.g. SI = 30 m) C stocks were between 70.2 and 71.1% larger than in SI = 22, across silvicultural scenarios (Table 5). The difference in C stock between PP and ST1, ST2 and ST3 was −3.2, 6.8 and 8.3 MgC ha^{-1} and −6.4, 30.2 and 35.1 for SI = 15 and 30 m, respectively, when compared with SI = 22 m.

At initial stand density of 750 trees ha^{-1} net C stock was reduced 9.3, 12.8, 16.2 and 17.7 MgC ha^{-1} for PP, ST1, ST2 and ST3, respectively, when compared to initial stand density of 1500 trees ha^{-1} (around 8.8 to 14.1% net loss). This reduction was explained principally by a decrease in slash pine production of 10.6, 10.1, 12.6 and 11.7 MgC ha^{-1} on the same silvicultural scenarios. The effect on woody products was not homogeneous, producing gains of 4.2 and 0.4 MgC ha^{-1}on PP and ST1, but reductions of 1.4 and 3.7 MgC ha^{-1} on ST2 and ST3, respectively. When initial stand density was stepped up to 2250 trees ha^{-1}, net C stock was increased 5.6, 7.5, 8.2 and 9.9 MgC ha^{-1} when compared to default initial stand density. On high initial stand density, even that *in situ* C stock was increased 8.4, 8.0, 9.5 and 8.8 MgC ha^{-1} on PP, ST1, ST2 and ST3, respectively, *ex situ* C stock was reduced 4.6, 2.6, 2.7 and 0.5 MgC ha^{-1} on the same silvicultural scenarios.

Under PP (unthinned) scenario, shortening the rotation length to 18 years, which corresponded to the biological rotation length for unfertilized stands (the time when mean annual increment equals current annual increment), net C stock was reduced down to 87.2 MgC ha^{-1}. This 16.9% decrease, if compared with default 22 years rotation, was partitioned in similar proportions for *in situ* and *ex situ* C stock. On the other hand, when rotation length was lengthened up to 35 years, net C stock was increased up to 133.2 MgC ha^{-1}. Most of this increment (66%) was due to *in situ* C storage.

With an assumed decay rate of 10%, C storage in forest floor and dead trees increased between 11.1 and 11.7 MgC ha^{-1}, corresponding, approximately to a 46.1% increment across silvicultural scenarios. When the decay rate was increased up to 20%, C stock in the dead pool was reduced between 29 and 31%, corresponding to

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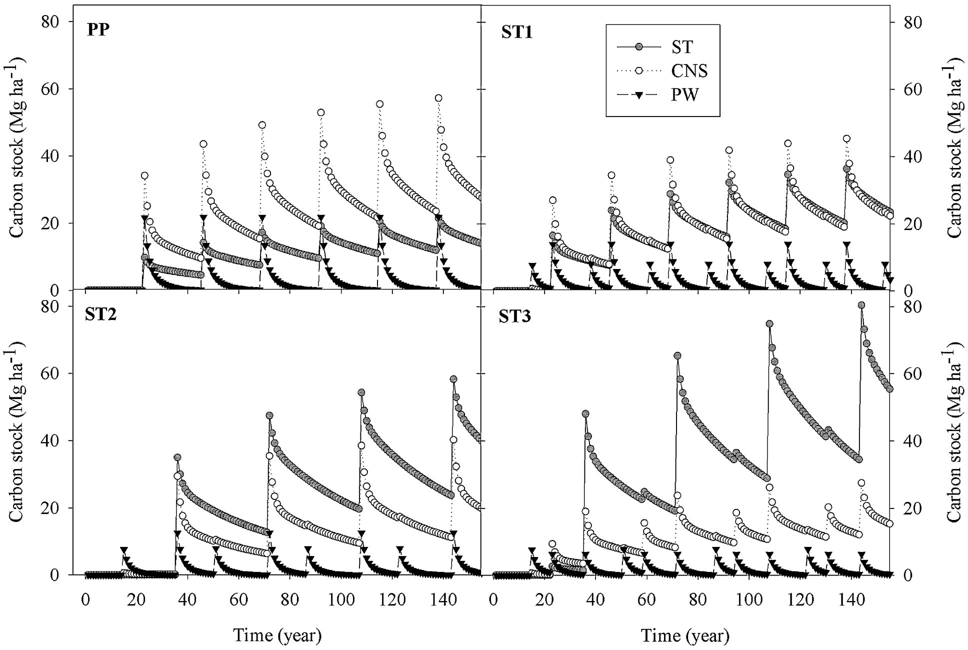

Fig. 4. Carbon stock in woody products for slash pine plantations under different silvicultural scenarios for a 150-year simulation period.

reductions between 7.4 and 7.7 MgC ha^{-1}, when compared to decay rate of 15%. Net C stock was increased up to 11.2% or reduced up to 7.3% when decay rate was 10 and 20%, respectively (Table 5). Under the low decay rate about 42, 44, 36 and 39% of *in situ* C stock was in dead pool on PP, ST1, ST2 and ST3, respectively, but under high decay rate, the proportion of *in situ* C in dead pool was reduced to 26, 28, 21 and 23%, with the same silvicultural scenarios.

Industrial efficiency of transformation of processing the harvested material to final products (timber or pulp) effects on C sequestration was assessed by changing conversion efficiency of the three major production lines analyzed (Table 3). There was a small impact on net C sequestration when conversion efficiency was increased or decreased by 5% (Table 5). Carbon storage in ST and CNS increased or decreased by 7.7% and C storage in PW (which corresponds between 1.7 and 3.6% of net C stock) changed by 8.6% when conversion efficiency was increased or decreased by 5%.

When average lifespan of ST was increased by stepping up product proportion in long-lived class (half-life 50 years, Table 3) from 50 to 75%, net C stock was increased mainly in sawtimber-oriented silvicultural schemes, increasing net C stock in 2.2, 3.8, 5.8 and 8.0% for PP, ST1, ST2 and ST3, respectively (Table 5). On the other hand, decrements of the same magnitude but in opposite direction was

Table 5
Sensitivity analysis for selected parameters under different silvicultural scenarios.

Parameter	Value	PP		ST1		ST2		ST3	
		MgC ha^{-1}	Δ%	MgC ha^{-1}	Δ%	MgC ha^{-1}	Δ%	MgC ha^{-1}	Δ%
Site Index (m)	15	62.5	−40.5%	59.2	−41.5%	69.2	−43.5%	70.7	−43.9%
	30	179.4	70.9%	172.9	70.9%	209.5	71.1%	214.4	70.2%
Planting density (trees ha^{-1})	750	95.7	−8.8%	88.4	−12.7%	106.3	−13.2%	108.3	−14.1%
	2250	110.6	5.4%	108.7	7.4%	130.6	6.7%	135.9	7.8%
Rotation length (years)	18	87.2	−16.9%	–	–	–	–	–	–
	35	133.2	26.9%	–	–	–	–	–	–
Decay rate (%)	10	116.7	11.2%	112.7	11.3%	133.6	9.1%	137.5	9.1%
	20	97.5	−7.1%	93.8	−7.3%	115.1	−6.0%	118.3	−6.1%
ST Conversion Efficiency (%)	60	104.3	−0.6%	100.1	−1.1%	120.6	−1.5%	123.4	−2.1%
	70	105.6	0.6%	102.3	1.1%	124.3	1.5%	128.6	2.1%
CNS Conversion Efficiency (%)	60	103.5	−1.4%	100.0	−1.1%	121.5	−0.8%	125.1	−0.7%
	70	106.4	1.4%	102.4	1.2%	123.4	0.8%	126.9	0.7%
PW Conversion Efficiency (%)	53	104.7	−0.2%	100.9	−0.2%	122.3	−0.1%	125.8	−0.1%
	63	105.2	0.2%	101.4	0.2%	122.6	0.1%	126.1	0.1%
ST lifespan long life class %	25	102.6	−2.2%	97.3	−3.8%	115.3	−5.8%	115.9	−8.0%
	75	107.2	2.2%	105.1	3.8%	129.6	5.8%	136.1	8.0%
CNS lifespan long life class %	0	99.6	−5.1%	96.9	−4.2%	118.0	−3.6%	122.0	−3.1%
	50	110.3	5.1%	105.5	4.2%	126.9	3.6%	130.0	3.1%
PW lifespan medium-short class %	0	103.9	−1.0%	100.1	−1.1%	121.8	−0.6%	125.3	−0.6%
	67	106.0	1.0%	102.3	1.1%	123.2	0.6%	126.7	0.6%

Note: Net carbon stock (MgC ha^{-1}) is the average for first 5 rotations and Δ% is the percentage deviation from default parameter values used (Site index = 22 m; Planting density = 1500 trees ha^{-1}; Rotation length = 22 years; Decay rate = 14%; ST, CNS and PW conversion efficiency = 65, 65 and 58%, respectively; ST in long life class = 50%; CNS in long life class = 25%; PW in medium-short life class = 33%).

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observed when ST proportion on long-lived lifespan products was stepped-down to 25%. The impact of CNS half-life in net C stock was similar across silvicultural scenarios (between 3.1 and 5.1% difference, Table 5). An absolute change of about 4.5 MgC ha^{-1} was observed when CNS proportion in long-lived lifespan products was increased by 50% or down to 0%. Paper product lifespan had a small effect on net C storage (Table 5). When all PW products were set to have a half-life of 1 year, net C stock was reduced between 0.7 and 1.1 MgC ha^{-1}.

4. Discussion

It is widely assumed that forests and forest management have the potential to play an important role in the active mitigation of atmospheric CO_2 (Schlamadinger and Marland, 1996; Sedjo et al., 1997; Marland et al., 1997; Canadell and Raupach, 2008; Malmsheimer et al., 2008). Additionality is a key attribute that must be considered in the design of any C offset project (Nabuurs et al., 2007). Additionality is the principle that in order for offset projects to have a real effect on atmospheric CO_2, the project must sequester more CO_2 than would have been sequestered in the absence of the project (Malmsheimer et al., 2008). For forestry projects in which trees are planted on previously unforested areas (afforestation) or in areas unforested for some period of time (reforestation), additionality is usually easily established, since the forest use in most cases sequesters substantially more CO_2 than the previous, non-forest landcover. In contrast, the application of forest management approaches to increase C sequestration in existing forests have at times been met with skepticism (Sawyer, 1993; Intergovernmental Panel on Climate Change, 2002; Niesten et al., 2002; Schwarze et al., 2002; Kallbekken et al., 2007), in part because the effects of forest management are thought to be too small or subtle to quantify (Chomitz, 2000). The research reported here is intended to address this problem by developing a robust modeling framework for evaluating forest management effects on C sequestration in an important forest type, and to use this tool to assess a range of management scenarios and their effects on C sequestration in forests and in products. While the simulations were based on data and models for *Pinus elliottii*, it is likely that the principles apply to a range of managed forests in North America.

When the simulated values were compared with reported accumulated C stock at age 24 years for the previous plantation on the same site, our estimation fits within the interval reported by Clark et al. (2004). The authors determined an average of 108.7 MgC ha^{-1} with a standard deviation of 11.7 MgC ha^{-1}, while at the same age this study determined a C stock of 113.8 MgC ha^{-1}. The same authors reported average C content in forest floor and coarse wood of 25.9 and 46.7 MgC ha^{-1} before and after clear cutting at age 24, with standard deviations of 6.5 and 17.6 MgC ha^{-1}, respectively; our simulations estimated values of 18.7 and 53.0 MgC ha^{-1}, respectively (data not shown). Shan et al. (2001) reported mean values of 3.2, 26.0 and 120.3 Mg ha^{-1} for understory, forest floor and total tree aboveground biomass of fertilized slash pine stands at age 17. Although variation was not reported, these values are also consistent with our estimates of 2.2, 26.7 and 126.4 Mg ha^{-1} for the same compartments at similar SI and stand density (data not shown). These results support the robustness of the model and give a solid basis for model application to C sequestration analysis.

The bulk of experimental evidence suggests that the impact of forest management on net C balance of *in situ* C depends largely on C storage in vegetation rather than soil (Johnson et al., 2002; O'Brien et al., 2003). Several authors have reported little or no impact of forest management (including stand development and harvests) on C storage in soil in a range of ecosystems, including *Eucalyptus regnans* and *Pinus radiata* (O'Brien et al., 2003), *Pinus elliottii* (Gholz and Fisher, 1982; Harding and Jokela, 1994; Johnson et al., 2002), and *Pinus strobus* (Knoepp and Swank, 1997). Shan et al. (2001) also reported no effect of fertilization or weed control on total soil profile C content (up to 1 m) on 17-year-old slash pine. Johnson and Curtis (2001), using a meta analysis approach, concluded that forest harvesting had negligible effect on soil C content. On the other hand, land use change can alter C content in soil, as increase in soil C when crop lands are reverted to forest or reductions in soil C when grasslands are converted to plantations in high rainfall sites (Guo and Gifford, 2002).

Emissions associated with silvicultural activities, including transportation of logs to the mill, are small compared to the magnitude of net C sequestration, accounting between 2.1 and 2.3% of gross C stock. Our estimates of transportation cost between 1.15 and 1.55 MgC ha^{-1} are in agreement with those reported by Berg and Karjalainen (2003) of 1.04 and 1.28 MgC ha^{-1} for North-European coniferous plantations. In terms of total C cost of silviculture the same authors reported between 5.0 to 5.6 MgC ha^{-1} (estimated from total annual CO_2 emissions from forest operations), while in our study we estimated total C cost of silviculture between 2.3 and 3.0 MgC ha^{-1}. The lower C cost of silvicultural management of slash pine plantations could be related to the lower number of interventions and the higher productivity of slash pine forests. These results emphasize the large "C return on investment" associated with silviculture in these systems. In other words, the sometimes very large increases in productivity associated with management intensification (e.g. Jokela and Martin, 2000; Borders et al., 2004; Jokela, 2005) is an order of magnitude larger than the C "invested" or emitted by the silvicultural activities themselves. This is of course distinct from the economic return on investment associated with silvicultural treatments and C payments, a topic beyond the scope of this paper but developed elsewhere (Stainback and Alavalapati, 2002; Alavalapati and Stainback, 2005; Boyland, 2006; Dwivedi et al., 2009).

Canadell and Raupach (2008) pointed out that increasing forest C density through stand-scale management and landscape-scale strategies, such as longer harvesting cycles, is one of the four major strategies to achieve net C sequestration. In our study we estimated net increments of 28 and 21 MgC ha^{-1} in average C stock of first 5 rotations when rotation length was increased from 22 to 35 years on unthinned and thinned stands, respectively. While the *in situ* C pool in slash pine accounts for most of this net increment, C in ST increased from 8 and 14 to 23 and 24 MgC ha^{-1}, on unthinned and thinned stands, respectively. Other reports also sustain these results as Liski et al. (2001), who indicated that longer rotations are favorable for C sequestration in Scots pine in Finland, with around 12 MgC ha^{-1} increase in net C stock when rotation length was increased from 60 to 90 years. Harmon and Marks (2002) also concluded that increasing rotation length increased the total amount of C stored in Douglas-fir. They reported around 77% increase in C stock when rotation was increased from 40 to 60 years under high utilization, severe slash burning system.

Thinning effects on net C stock depended on intensity and timing of intervention. On short rotation scenarios (22 years) thinning at age 14 slightly reduced net C stock (3.6% reduction). This was mainly the net effect of 5 and 4 MgC ha^{-1} reduction in slash pine biomass and CNS production, but an increase of 6 MgC ha^{-1} in ST partially counteracted that loss. On long rotation scenarios (35 years) thinning at age 14 reduced net C stock by 11 MgC ha^{-1}. This decrease was mainly explained by a reduction of 9 MgC ha^{-1} of *in situ* C. As the fate of volume extracted in thinnings is primarily pulpwood, the short life of PW was reflected on negligible effects on C sequestration. When a second thinning was carried out at age 22 on long rotation scenario, there was a net increase in C stock of 4 MgC ha^{-1} compared with 1 thinning scenario, mainly due to changes in diameter classes on slash pine trees, promoting higher

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proportion of long-lived products. Working on mixed coniferous stands in Finland, Garcia-Gonzalo et al. (2007) reported net reduction between 25 and 33 MgC ha^{-1} in trees and net increase between 30 and 45 MgC ha^{-1} in harvested timber. The effect of thinning on C stock depended on thinning intensity and timing.

Site productivity was the major factor driving modeled C sequestration in slash pine stands, although the magnitude of change interacted with management scenario. For example, when sites with average productivity (e.g. SI = 22 m) and low productivity (e.g. SI = 15 m) were compared under the PP scenario, both *in situ* and *ex situ* C stocks declined about 50%. Under the more intensive ST3 scenario, though, *ex situ* stocks in high SI stands were 56% higher than in stands with lower productivity. For low SI sites, the proportional reduction in C stock in ST product classes as a fraction of all *ex situ* stocks was 40, 62, 81 and 90% for PP, ST1, ST2 and ST3, respectively, indicating that long rotation sawtimber-oriented silvicultural schemes are not suitable for C storage on low productivity sites. Conversely, on higher productivity sites (e.g. SI = 30 m) the increase in net C stock was largely in *ex situ* rather than *in situ* C pools, averaging relative increment of 59 and 62% for *ex situ* pools PP and ST3, respectively, if compared with SI = 22 m. In this case, the increment in *ex situ* C was largely due to the ST product class, accounting for more than 90% of *ex situ* C increment across silvicultural schemes. These results indicate that on high productivity sites, silvicultural schemes that promote sawtimber products are better suited for increasing C storage, and also emphasize the importance of considering both *in situ* and *ex situ* carbon pools when considering the C offset potential of different management scenarios.

Planting at low initial plant density (e.g. 750 trees ha^{-1}) stored less C than planting 1500 trees ha^{-1} and this net reduction was larger than the benefit of increasing initial plant density up to 2250 trees ha^{-1}. Under low initial plant density on PP regime, the decrease of *in situ* C was partially counteracted by the increase on *ex situ* C (largely ruled by ST production) but on ST2 and ST3 regimes both C pools were decreased. Diameter class changes due to sawtimber-oriented silviculture are not sufficient to counteract the lower number of trees in low initial planting density stands. Under high initial plant density, *in situ* C was positively and very homogeneously affected across silvicultural regimes, but *ex situ* C was decreased, affecting more PP rather than ST3 regime. Changes in diameter classes down to CNS and PW products explain the negative effect on *ex situ* C.

Litter decomposition rate can be affected by changes in litter quality and/or changes in environmental conditions as soil temperature and moisture (Anderson, 1991), so is probable that environmental changes as a consequence of global warming will lead to changes in litter decomposition rates on slash pine plantations. In our study when we simulated that combined effect by reducing decay rate down to 10% or stepping it up to 20%, it was observed that under lower decay rate C stock in dead pool (forest floor + dead trees) was increased between 11.1 and 11.7 MgC ha^{-1} across silvicultural regimes, and under high decay rate scenario, C stock in dead pool was decreased between 7.4 and 7.7 MgC ha^{-1}. Changes in litter element ratios and chemical content might also affect the optimal fertilization regime.

Extending lifespan of PW products has marginal effect on C stock. Similar response was reported by Gundimeda (2001) for lengthening by 2 years the lifespan of short and medium lifespan products in India. In our study, when CNS and ST lifespan was increased in long life class, there was between 2.2 and 8.0% increment in net C stock. Harmon et al. (1996), working on Oregon and Washington forests, reported that increasing long-term structure replacement from each 100 to each 50 or 200 years decreased 9.9% or increased 6.6% C store in woody products, respectively. Gundimeda (2001) pointed out that by increasing lifespan of long life (50 years) products, C can be stored over a period equal to the time needed to grow the timber for those products. We concluded that changes in utilization of sawtimber products to long-term structures can lead to enhancements in C sequestrations. Although reporting of C stocks in wood products is optional under the United Nations Framework Convention on Climate Change (UNFCCC), the enhancement of such stocks could provide significant GHG emissions offsets.

One component not considered in this analysis is forest products stored in landfills. Micales and Skog (1997) have reported that the decay of solid wood in landfills is negligible and for paper products it does not exceed 30%. This implies that wood waste deposited into landfills may form a very important C sink in the life cycle of wood products. On the other hand, as Pingoud et al. (2001) pointed out, if conditions in landfills became favorable for decomposition of wood product C stocks, the resulting methane emissions could have a significant net GHG impact.

Carbon sequestration is not currently the primary driver for management of most forest stands. Forest C sequestration should be considered as an element of the portfolio of goods and services which are derived from forests. In the presence of a C market, these services will in some cases have market value, and so C offset values will likely be considered alongside traditional forest product values in economic assessments of management alternatives. In the absence of viable C offset markets, forest C sequestration will remain an important non-market service, and as such will continue to be an important ecosystem management consideration for the foreseeable future.

5. Conclusion

We developed a model to account for C stock and fluxes in slash pine plantation ecosystems in North Florida. The robustness of the model was tested through a comparison of different model outputs against a wide range of C measurements. This model accurately simulates C storage in slash pine plantations from establishment to maturity over a wide range of site quality. Using the model to evaluate the effect of silvicultural management on C sequestration of slash pine plantations, we found that (i) Site productivity, influenced by silviculture, is the major factor driving C sequestration in slash pine stands; (ii) On high productivity sites, silvicultural schemes that promote sawtimber-type products are more suitable for increasing C storage; (iii) Increasing rotation length increases C storage; (iv) Although *in situ* C pool in slash pine accounts for most of this net C storage, C in sawtimber products can account for 26% of total C storage on long rotation-thinned stands; (v) Emissions associated with silvicultural activities are small compared to the magnitude of net C stock. This slash pine plantation C sequestration model, based on transparent empirical and biological relationships, is appropriate for use in regional C stock assessments or for C credit verification.

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