

Processes influencing soil carbon storage following afforestation of pasture with *Pinus radiata* at different stocking densities in New Zealand

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Abstract. Since 1992, afforestation with *Pinus radiata* D. Don in New Zealand has led to the establishment of over 600 000 ha of new plantation forests, about 85% of which are on fertile pastures used previously for grazing sheep and cattle. While this leads to rapid accumulation of carbon (C) in vegetation, the effects of afforestation on soil C are poorly understood. We examined key soil C cycling processes at the (former) Tikitere agroforestry experimental site near Rotorua, New Zealand. In 1973, replicated stands of *P. radiata* (100 and 400 stems/ha) were established on pastures, while replicated pasture plots were maintained throughout the first 26-year rotation. In 1996, soil C and microbial biomass C in 0–0.10 m depth soil, *in situ* soil respiration and net N mineralisation, and soil temperature were lower in the forest than in the pasture, and tended to decline with increasing tree-stocking density. In the 400 stems/ha stands, mineral soil C (0–0.50 m depth) was lower than in the pasture (104 and 126 Mg C/ha, respectively; $P < 0.01$). Carbon accumulation in the forest floor during the first rotation of these forest stands was 12 Mg C/ha. Using the Rothamsted soil C model (Roth-C), we examined how changes in plant C inputs following afforestation might lead to changes in soil C content to 0.30 m depth. Steady-state pasture inputs of 9.0 Mg C/ha.year were estimated using Roth-C; these C inputs were assumed to decrease linearly during the first 12 years following tree establishment (until canopy closure). Below-ground C inputs in the forest were estimated using steady-state relationships between litterfall and soil respiration; these inputs were assumed to increase linearly between years 1 and 12, after which they remained constant at 1.53 Mg C/ha.year until harvest. Measured changes in soil C (0–0.30 m) during the first rotation, in conjunction with the below-ground inputs, were used to estimate above-ground inputs (as a proportion of total litterfall [3.81 Mg C/ha.year]) to the soil. Our results suggest 10% of litterfall C over one rotation actually entered the mineral soil. Using these results and estimates of additional C inputs to the soil from harvest slash and weeds following harvest, we found mineral-soil C stocks would continue to decline during second and third rotations of *P. radiata*; the magnitude of this decline depended in part on how much slash enters the mineral soil matrix. We confirmed our modelling approach by simulating soil C changes to within 8% over 19 years following afforestation of pasture at another previously studied site, Purukohukohu. Whether afforestation leads to an increase or decrease in mineral-soil C may depend on previous pasture management; in highly productive pastures, high C inputs to the soil may maintain soil C at levels that cannot be sustained when trees are planted onto these grasslands.

Additional keywords: carbon cycling, nitrogen cycling, plantation, forest, land-use change, forest management.

Introduction

Afforestation of agricultural land (arable or pastoral), through abandonment or establishment of plantation forests, is a major global land-cover change. In New Zealand, the area of plantation forests dominated by *Pinus radiata* D. Don is about 1.8 million ha (MAF 2003). From 1992 to 2002, new plantation forest plantings ranged between 32 000 and 100 000 ha/year; total area planted was 618 000 ha (MAF 2003). Over half these new plantings occurred on abandoned pastures. Likewise in Australia, about 220 000 ha

of agricultural land was converted to forest vegetation between 1991 and 1995 (Barson *et al.* 2000). These land-cover changes have important consequences under the Kyoto Protocol where countries can claim credits for increased C storage resulting from afforestation. While vegetation-C stocks obviously increase with afforestation, changes in soil C are poorly understood (Paul and Polglase 2004), yet must be reported in accordance with international best practice guidelines along with changes in vegetation-C storage (IPCC 2000).

To meet its obligations under the Framework Convention for Climate Change and the Kyoto Protocol, New Zealand has adopted a net CO₂-reductions policy based in part on increased planting of plantation forests (primarily *P. radiata*) as an offset against rising CO₂ emissions (MfE 1997). Current C budget models used to calculate changes in C storage in plantation forests include increased C storage in the forest floor during stand development, but assume no changes in mineral soil C (Ford-Robertson 1997). Carbon storage in the top 0.30 m of mineral soil varies greatly as a function of soil type (Tate *et al.* 1997), but on an area basis could easily equal the amount of C stored in plantation-forest vegetation after harvest removals are subtracted (Maclaren 1996). Small changes in soil C with afforestation could, therefore, affect net C accumulation in the whole forest ecosystem. Previous studies have observed decreases in mineral soil C storage in surface layers after one rotation of plantation forests in New Zealand (e.g. Alfredsson *et al.* 1998; Scott *et al.* 1999), but these results are equivocal (e.g. Giddens *et al.* 1997).

Conversion of grazed pasture to plantation forests is accompanied by major changes in the quality, quantity, and distribution (spatial and temporal) of C inputs to the soil (Tate 1987). In pasture ecosystems, most C inputs are below ground via root exudation and fine-root turnover. These inputs are readily decomposable, leading to high microbial activity in the rhizosphere of pasture soils (Tate 1987; Sagar *et al.* 2000). Grazing also influences both the amount and distribution of detrital C inputs in grasslands (Holland and Detling 1990). In contrast, a greater proportion of detrital C inputs in plantation forests come from above-ground litterfall. These above-ground inputs are likely to have higher lignin concentrations (Scott and Binkley 1997), but lower N concentrations, than either above- or below-ground inputs in pasture ecosystems. In addition, because they are located on the soil surface, these inputs may decompose more slowly than grassland detritus because of frequent desiccation. Over a range of forest ecosystems at steady-state (i.e. detrital C inputs equal soil CO₂-C losses), the quantity of C contained in above-ground litterfall is roughly 2.5-fold lower than that contained below-ground (Raich and Nadelhoffer 1989). Below-ground C inputs could, therefore, also be appreciable (e.g. Giardina and Ryan 2002).

Many countries have begun developing national C monitoring systems to quantify and report soil C stocks (Scott *et al.* 2002) and changes in these stocks associated with land-use change (e.g. Paul *et al.* 2003; Tate *et al.* 2005). Some of these systems have included simulation models to predict future soil C changes (Paul *et al.* 2003). Among the more difficult processes to simulate in these models have been changes in litter decomposition and incorporation of litter C into the mineral soil (Halliday *et al.* 2003). These processes are being represented in highly parameterised, or more empirical, models (e.g. Paul and Polglase 2004). However, for C monitoring systems to be applied in different countries

and with varying land-use change scenarios, simpler, less data-intensive approaches are needed to quantify rates of litter decomposition and C inputs to the mineral soil. The Roth-C soil C model (Coleman and Jenkinson 1996) uses 3 litter quality 'classes' based on the decomposition rates of organic residues of differing composition. The model does not, however, simulate the transfer of surface litter C into the mineral soil.

Here, we examine changes in biogeochemical processes associated with conversion of pasture to plantation forest (*P. radiata*) that influence soil C turnover and storage. Using a replicated agroforestry experiment at Tikitere, where *P. radiata* was planted at different densities into pre-existing pasture, we tested the hypothesis that soil C storage increases with conversion of pasture to pine forest largely because of a decline in nutrient cycling and soil C turnover rates. We used our results, and a soil C simulation model (Roth-C model; Coleman and Jenkinson 1996), to explore how changes in soil C inputs and soil temperature might influence soil C storage following afforestation. We tested our modelling approach using data from a previously studied pasture/pine conversion (Purukohukohu [Puruki]); Ross *et al.* 1999; Parfitt *et al.* 2003).

Materials and methods

Site properties

We examined changes in soil C accumulation and turnover, and rates of nutrient cycling, following afforestation at the Tikitere agroforestry experiment near Rotorua, New Zealand (38° 04' S, 176° 20' E). This experiment was established in 1973 to examine forest and pasture productivity in agroforestry systems, and the relationship between livestock carrying capacity and *P. radiata* density. Before planting, the site was used for pastoral grazing for about 40 years (M. F. Hawke, pers. comm.). Trees were established at densities of 250–2000 stems/ha, then gradually thinned to final densities of 50, 100, 200, and 400 stems/ha up until 1981. The site also contained pasture with no trees (0 stems/ha). The experimental design was a randomised complete block, with blocks (4) stratified to represent topographic variation around the site (low to high). Plot size was 2 ha, with each plot buffered by a 28-m surround planted at the same density as the plot. Pasture species disappeared from the 400 stems/ha treatment by the time the trees reached 13 years of age (Hawke 1991), whereas pasture production was still 40–50% of that in the open pasture in the 100 stems/ha treatment in 1992 (Knowles *et al.* 1995). In the New Zealand Soil Classification, soils are Typic Orthic Pumice (Hewitt 1998), and in the US Soil Taxonomy (Soil Survey Staff 1998) are Typic Udivitrands. Mean annual rainfall is 1490 mm; mean daily temperature is 7.3°C in July and 17.8°C in February. Tikitere presented an ideal site to examine the effects of trees on soil C dynamics, as the different densities of trees had, through time, gradually outcompeted the grass, creating a mosaic of plots with different C inputs from the pasture and *P. radiata*. Further details on the establishment and maintenance of this experiment can be found in Hawke and O'Connor (1993) and Hawke and Knowles (1997). The forest was harvested in 1999.

Soil and forest floor sampling

Sampling of the mineral soil in the open pasture, and mineral soil and forest floor (L and FH material) from the 100 and 400 stems/ha treatments, was undertaken in February 1996. Only 3 of the 4 field blocks were sampled, as the fourth was too steep to permit access with

the truck-mounted Giddings soil corer (Giddings Corp., Fort Collins, CO). Mineral soil cores (6.35 cm diameter, 20/plot) were collected along two roughly diagonal transects across each plot to a depth of 0.20 m. Each core was split into 0–0.10 and 0.10–0.20 m depth increments. Measurements of core length indicated little compaction during coring. At alternating sampling locations (total of 10/plot), an additional core was taken to 0.50 m depth. All cores taken at the same depth within a plot were composited for subsequent analyses. We sampled forest floor material at 10 locations along transects in each plot using a 0.1225 m² quadrat.

Soil properties

Bulk density of forest floor and mineral soil was determined by weighing the entire composite, field-moist sample, subsampling, then drying for 24 h at 105°C. Subsamples of the composite, field-moist samples of mineral soil were wet-sieved (2 mm), removing large quantities of pumice. The deepest layer (0.20–0.50 m) was passed through a 5.6-mm sieve. Pumice density was estimated by submerging material retained on the 2-mm sieve in water and recording displacement. The same material was then dried, weighed, and discarded. The field-moist forest floor material was likewise sieved (5.6 mm) before analyses. Sieved subsamples from all sampling depths were air-dried, ground, and analysed for total C and N on a LECO FPS-2000 CNS Analyzer (LECO Corp., St. Joseph, MI).

Moisture content was determined by drying samples at 105°C for 24 h. Soil pH was measured on a 2.5:1 v/w mixture of water and soil (Blakemore *et al.* 1987). Microbial C and N were measured using the fumigation–extraction procedure as described by Ross *et al.* (1999). The experimentally determined k_{EC} -factor (the ratio of extractable-C flush/microbial C) in 0–0.10 m depth samples averaged 0.35; this factor was used throughout for converting extractable C flush to microbial C. The k_{EN} -factor used for converting extractable N flush to microbial N was 0.45 (Jenkinson 1988).

Laboratory C and net N mineralisation rates

Soil CO₂ production was measured under controlled laboratory conditions at 25°C using 20.0 g of mineral soil or 10.0 g forest floor material, at 60% of water-holding capacity (WHC). The samples were incubated in 125-mL polypropylene cups in 1.8-gas-tight jars containing 10 mL water to maintain humidity. The jars were sealed with lids fitted with a septum. We measured CO₂ production over 56 days by periodically sampling headspace CO₂ concentrations and injecting a 1-mL sample into a gas chromatograph (Carle 8700) equipped with a thermal conductivity detector. The jars were then opened and sample containers weighed to monitor water content. When necessary, deionised water was added to maintain constant soil moisture conditions. The jars were then flushed with ambient air, re-sealed, and returned to the incubator.

We measured aerobic net N mineralisation by incubating 10.0 g of mineral soil or 5.0 g of forest floor material at 60% of WHC in 125-mL polypropylene cups. Here, the cups were covered with polythene and placed in a large plastic tray containing water, which was then placed inside a plastic bag to reduce moisture loss. We monitored soil water content throughout the 56-day incubation at 25°C, but no water additions were necessary. Net N mineralisation was measured at 56 days by extracting the samples with 100 mL of 2 M KCl, shaking for 1 h, then filtering through MFS 5C filter paper (Whatman 40 equivalent). Extractable NO₃[−]-N and NH₄⁺-N were measured colourimetrically on an AutoAnalyzer II (Blakemore *et al.* 1987).

In situ net N mineralisation

Because of the long distance between our field site and home laboratory, we were unable to measure annual *in situ* net N mineralisation. We did, however, measure net N mineralisation on 2 occasions by the buried bag technique (Eno 1960), using 8 cores (6.35 cm diameter;

0–0.10 m deep) spaced at 10-m intervals in each plot and incubated over 6-week intervals between February and April 1996 (summer), and between November and December 1996 (spring). During the first sampling interval, *in situ* net N mineralisation was measured in the pasture and 400 stems/ha treatments, and during the second interval in the pasture, 100, and 400 stems/ha. At the beginning and end of the incubation period, soil subsamples (10 g field-moist) were extracted with 100 mL of 2 M KCl, shaken for 1 h, then filtered (MFS 5C). Nitrate-N and NH₄⁺-N were measured colourimetrically on an AutoAnalyzer II (Blakemore *et al.* 1987).

Above-ground litterfall and in situ soil respiration

Above-ground litterfall (100, 200, and 400 stems/ha treatments) was measured in all 4 blocks at approximately monthly intervals over 24 months starting in January 1996. In each plot, 8 litter traps (0.126 m²) were located at 5-m intervals along a transect located near the centre of each plot. Samples from each trap were collected, dried at 60°C, and weighed for total mass. Needles and branches were weighed separately. Samples from the 8 traps were then combined, ground in a UDY cyclone mill (1.0 mm mesh) (UDY Corporation, Fort Collins, CO), and analysed for C and N on the LECO FPS-2000 CNS Analyzer.

Soil CO₂ production *in situ* was measured twice; once in February 1996, and again in November 1996, using a portable infrared gas analyser (PP Systems, Hertfordshire, UK). CO₂ flux was measured at 10 locations in each plot along an approximately 100-m transect; soil temperatures (0.05 m depth) were recorded in conjunction with each soil respiration measurement.

Soil temperature

In addition to the above soil temperature measurements, we monitored soil temperatures between February 1996 and March 1997 using HOBO temperature loggers with external thermistor cables (Onset Computer Corporation, Pocasset, MA). Temperature sensors were inserted laterally into an exposed soil face at 0.02 and 0.05 m depth, after which the datalogger and sensor were buried. Only 1 plot for each of the treatments (0, 100, 400 stems/ha) was used for these measurements.

Statistical analysis

Treatment effects on soil C and N properties and litterfall were determined using *t*-tests for FH material and the Generalized Linear Model (GLM) procedure with Pairwise Comparisons in SYSTAT (SYSTAT 1996) for mineral soil. Soil depth increments were analysed independently. Both block number and treatment were included as factors in the model; this is a legitimate model given that interaction terms appeared to be unimportant. Block effects were not assessed in the analysis. When treatment differences were significant ($P < 0.05$), treatment means were compared using Fisher's l.s.d. test.

Modelling soil C changes

Simulating long-term effects of afforestation on soil C using Roth-C

We used the Roth-C model (Coleman and Jenkinson 1996) to examine the sensitivity of soil C levels in the top 0.30 m of the mineral soil to changes in C inputs and soil temperature following afforestation in the 400 stems/ha treatment. First, we parameterised the model using soil and litterfall measurements from Tikitere (first rotation). Second, we tested our parameterisation at a paired pasture/pine site with extensive soils information (Puruki; Ross *et al.* 1999; Parfitt *et al.* 2003). We then used the model to explore soil C changes at Tikitere beyond the first rotation of *P. radiata*.

Simulation scenarios for Tikitere—input estimates and temperature changes

We estimated mineral-soil C inputs during the first rotation of *P. radiata* at Tikitere in the following way. Pasture inputs before forest planting

were simulated by Roth-C assuming a steady-state relationship between inputs (net primary production) and soil C stocks to 0.30 m (Jenkinson *et al.* 1992); this steady-state assumption appears to be reasonable for established New Zealand pastures (Tate *et al.* 2003). Pasture inputs were reduced linearly to zero between years 1 and 12 (Table 1), which coincided with canopy closure and elimination of grasses in the 400 stems/ha treatment (Knowles *et al.* 1995). Above-ground C inputs from the trees to the soil were based on litterfall measurements (Fig. 1) of 3.8 Mg C/ha.year in the 400 stems/ha treatment at 22 years. Our results suggest little interannual variation in litterfall, so we assumed that above-ground inputs would remain constant after canopy closure; this assumption is supported by litterfall measurements from 1983 to 1988 in the 200 stems/ha treatment (M. F. Hawke, unpublished data). Below-ground C inputs from *P. radiata* were assumed to increase linearly from year 5 to year 12, and to be constant thereafter until harvest (Table 1). From year 12, estimates of below-ground C inputs were based on steady-state relationships between litterfall, below-ground C inputs, and soil respiration (Raich and Nadelhoffer 1989). Based on the relationship shown in their Fig. 2 (total root C allocation = 1.92 (litterfall C) + 130), we estimated root C allocation to be 862 g/m².year (8.62 Mg/ha.year) for the average annual litterfall of 3.81 Mg C/ha.year. Soil respiration, estimated as the sum of root C allocation and litterfall C, was therefore 12.43 Mg C/ha.year. Assuming autotrophic respiration is 57% of total respiration (Ross *et al.* 2001),

Table 1. Estimated soil C inputs during growth of *P. radiata* in the 400 stems/ha treatment at Tikitere for simulating soil C changes following afforestation

The trees were harvested in year 26 and second-rotation trees planted in year 27

Year	Pasture	Pine ^A	Pine ^B	Pruning	Weeds ^C	Harvest ^D
0	9.0	0	0	0	0	0
1	8.3	0.12	0	0	0	0
2	7.6	0.24	0	0	0	0
3	6.9	0.35	0	0	0	0
4	6.2	0.47	0	0	0	0
5	5.5	0.59	0	0.46	0	0
6	4.8	0.71	0.05	0	0	0
7	4.1	0.82	0.10	0	0	0
8	3.5	0.94	0.16	0	0	0
9	2.8	1.06	0.21	0	0	0
10	2.1	1.18	0.27	0	0	0
11	1.4	1.29	0.32	0	0	0
12	0.7	1.41	0.37	0	0	0
13–25	0	1.53	0.37	0	0	0
26	0	1.53	0.37	0	0	0.56
<i>Second rotation</i>						
27	0	0.12	0	0	1.0	0.56
28	0	0.24	0	0	1.0	0.56
29	0	0.35	0	0	1.0	0.56
30	0	0.47	0	0	1.0	0.56
31	0	0.59	0	0.46	1.0	0.56
32	0	0.71	0.05	0	0	0.56

^AInputs for 400 stems/ha, below ground.

^BInputs for 400 stems/ha, above ground.

^CAnnual weed inputs based on peak weed biomass measurements at a nearby harvested *P. radiata* plantation; includes above- and below-ground.

^DEstimated above- and below-ground inputs at harvest (total of 7.86 Mg C/ha from Table 2).

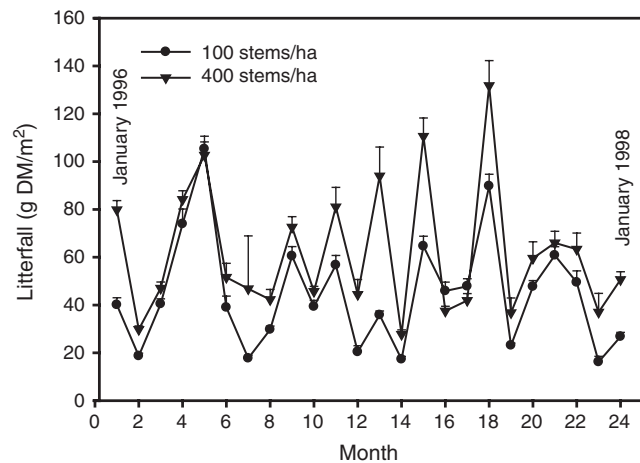


Fig. 1. Monthly (approximately) litterfall in the 2 *P. radiata* treatments.

heterotrophic respiration would be 5.34 Mg C/ha.year. From Eqn 1 in Raich and Nadelhoffer (1989), below-ground detritus production (P_b) is approximately equal to heterotrophic respiration (R_h) minus above-ground detritus production (P_a). P_b would consequently equal 5.34 minus 3.81, i.e. 1.53 Mg C/ha.year.

Not all litterfall reaches the mineral soil; decomposition occurs in the forest floor, and much of the litter C is respired directly back to the atmosphere. While some models simulate litter decomposition explicitly (e.g. Paul *et al.* 2003), we used the measured difference in soil C under pasture and pine, in conjunction with our estimate of below-ground inputs, to estimate C transfer from the forest floor to the mineral soil (the input to the mineral soil). As no specific site data were available for the various forest floor components at Tikitere, we were restricted to using a single rate of decomposition for litter.

First, we formulated the dynamics of litter C (L) by:

$$dL/dt = I_1 - k_1 L - k_2 L \quad (1)$$

where I_1 is annual litterfall (3.8 Mg C/ha.year), $k_1 L$ is the respiration rate of litter-C to CO_2 , and $k_2 L$ represents the transfer rate of C from the litter layer to the mineral soil. The forest floor mass (L) in the 400 stems/ha treatment contained 11.6 Mg C/ha, and was assumed to be at steady-state after 1 rotation. This assumption is supported by C contents found for the forest floor of other North Island *P. radiata* forests (Tate *et al.* 2001). Next, we represented mineral-soil C dynamics by:

$$dC/dt = I_2 \rho_1 + I_3 \rho_2 - k_3 AC \quad (2)$$

where $I_2 = k_2 L$ (above-ground inputs to the mineral soil), $I_3 = 1.53$ Mg C/ha.year (below-ground inputs to the mineral soil), and $\rho_i = (\kappa_i, 1 - \kappa_i, 0, 0)$, $i = 1, 2$ is a vector which represents the DPM:RPM ratios of inputs, with κ_i being the fraction of inputs that is decomposable plant material (DPM), and $1 - \kappa_i$ the fraction of inputs that is resistant plant material (RPM) (see Parshotam 1996). The expression $k_3 AC$ is from Roth-C, where k_3 is an annual average of rate modifying factors, and A is a matrix coefficient that describes mineral soil C turnover (Parshotam 1996). Therefore:

$$dC/dt = k_2 \rho_1 L + I_3 \rho_2 - k_3 AC \quad (3)$$

where k_2 in Eqns 1 and 2 is the difference between the rate constants describing the overall change in litter C (k) and respiration rate (k_1), i.e. $k_2 = k - k_1$, with $k = (3.81/11.6) = 0.33/\text{year}$ (annual litterfall/forest floor mass at steady-state). The known parameters are ρ_1 , ρ_2 , A , k_3 , k , and I_3 ; the unknown parameters to be estimated are k_1 and k_2 . The parameter k_1 can be estimated as 0.23 by substituting k_2

with $k - k_1$ in Eqn 3 and using the soil C data for year 26 (59 Mg C/ha) to 0.20 m depth. Therefore, $k_2 = 0.33 - 0.23 = 0.10$, which represents the fraction of total litterfall that is transferred annually from the forest floor to the mineral soil.

In addition to C inputs from the forest floor to the mineral soil due to litterfall, C inputs also come from pruning at year 5 (Knowles *et al.* 1995) (Table 1), slash produced at harvest, and weed growth during the first 5 years following harvest (Table 1). Carbon inputs due to pruning at age 5 were based on national average biomass statistics for a 5-year-old plantation of *P. radiata* (Maclaren 1996) and the site-specific thinning rate (Knowles *et al.* 1995). We assumed that 10% of the C in the pruning slash entered the mineral soil (based on our calculations for the litter layer). Similar national statistics (Maclaren 1996) were used to estimate slash production at harvest, using allometric data (Maclaren and Wakelin 1991) (Table 2) to derive estimates of foliage, branches, roots, stumps, and stemwood (merchantable and non-merchantable). For all the above-ground harvest-slash inputs, we assumed initially that 10% of this C entered the mineral soil, and the rest was released to the atmosphere as CO₂ (Table 2). Harvest-slash inputs were spread evenly over 14 years, based on observations at a second rotation site (see Ross *et al.* 2002) (Table 1). To simulate soil C changes for rotations beyond the first, similar *P. radiata* inputs were used as in the first rotation; we did not assume any changes in productivity in second and third rotations. Above-ground biomass of weeds after harvest was taken as 3.2 tC/ha (based on values from Parfitt *et al.* (2001) and Tate *et al.* (2006) at another clear-cut site); we assumed a similar below-ground weed C mass. Assuming above- and below-ground inputs were 10 and 20%, respectively, of the total weed C, we estimated

Table 2. Estimated C inputs to the mineral soil from harvest slash
Total stand biomass at 26 years based on national average biomass values (Maclaren 1996)

Material	Fraction of total harvested mass ^A	Mass of material (Mg/ha)	Allocation to mineral soil (%) ^B	C allocation to mineral soil (Mg C/ha) ^C
Live branches	0.128	51	10	2.6
Dead branches	0.028	11	10	0.6
Foliage	0.047	19	10	1.0
Stumps and coarse roots	0.055	22	10	1.1
Fine roots	0.007	2.8	10	0.1
Cones	0.007	2.8	10	0.1
Non-merchantable stemwood	0.06	24	10	1.2
Bark	0.038	15	0	0
Merchantable stemwood	0.4	160	0	0
Forest floor		23.2	10	1.16 ^D

^AValues for the fraction of the total harvested mass from Maclaren and Wakelin (1991). Note that they do not add to 100%, as they do not include 16% for the forest floor (site-specific values are used here for the forest floor).

^BThis is an estimate based on calculations of C transfer from forest floor to mineral soil. All merchantable stemwood and bark is assumed to have been taken from the site.

^CBased on 50% C in all tree components. The total C inputs would then equal 6.7 Mg C/ha plus 10% of the Tikitere forest floor C (7.86 Mg C/ha total).

^D10% of the forest floor mass (Table 5) is assumed to be an input after harvest.

weed C inputs to the mineral soil for 5 years after harvest would be about 1.0 tC/ha.year.

In addition to quantifying litter inputs, we also quantified differences in the quality of these inputs. In the Roth-C model, litter quality is represented using broad 'categories' based on the ratio of decomposable plant material to resistant plant material (DPM : RPM). Grassland inputs are generally assigned a value of 1.44, and for forests a much lower value of 0.25 (Jenkinson *et al.* 1991) is used because of the large amount of more resistant (to decomposition) compounds found in woody material (Tate *et al.* 1996). For our simulations, all tree inputs were assigned a DPM : RPM value of 0.25, and all pasture inputs a value of 1.44; weed inputs were assigned an intermediate value of 0.67. The only exception is that inputs from the young *P. radiata* forest (until 12 years) were assigned a value of 0.67, as inputs during this time contained little woody material. The same DPM : RPM ratios were assumed for above- and below-ground inputs for all plant types.

Air temperatures were used for the initial pasture steady-state simulation; these were found to roughly equal average daily soil temperatures under pasture (N. A. Scott, unpublished data). Average temperatures were about 3°C lower in the pine plantations compared with pastures (see Fig. 2), with this difference being assumed to apply from 1 year after canopy closure.

Other parameter values required for the Roth-C simulations were derived from previous work (e.g. Parshotam 1996). The size of the inert organic matter (IOM) pool was estimated at 6.6 Mg C/ha, based on established relationships between total C and IOM (Falloon *et al.* 1998, 2000); the soil clay content used was 5% (R. H. Wilde, pers. comm.).

First-rotation simulation scenario for Puruki

Given the similarity of the soils and land-use history at Tikitere and Puruki (both Typic Udivitrands; Parfitt *et al.* 2003), we used similar parameters and climate data (e.g. texture, litter quality, temperature) for both sites; the exceptions included site-specific values for precipitation, IOM, and C input scenarios. At Puruki, herbicide had been applied before planting the *P. radiata* seedlings (Beets and Brownlie 1987). The pasture C input was therefore assumed to be half (4.3 Mg C/ha) the steady-state input in the first year and zero from years 1–13. The loss of pasture inputs to the soil during the first few years of tree establishment was somewhat compensated for by inputs from weed growth. Weed inputs were based on measurements of weed biomass

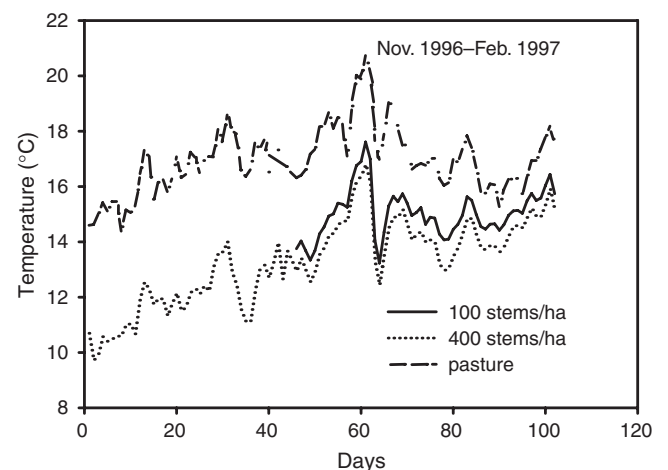


Fig. 2. Differences in mineral soil mean daily temperature at 0.05 m depth between the pasture, 100 stems/ha treatment, and 400 stems/ha treatment over 2 different time intervals. Actual readings were collected hourly and then averaged.

(Beets and Brownlie 1987); below-ground inputs were assumed to be twice above-ground inputs. Total weed inputs were taken as 0.3, 0.5, 1.2, 1.6, 1.4, 1.4, 1.4, and 0.5 Mg C/ha in years 1–8, respectively. Litterfall was similar at the 2 sites (3.8 at Tikitere v. 3.7 Mg/ha.year at Puruki (Parfitt *et al.* 2003)). We assumed similar below-ground inputs at both sites, although differences in some site characters (e.g. dead weed inputs at Puruki) could have influenced this assumption. Pruning inputs were estimated similarly to those at Tikitere, and amounted to 0.46 Mg C/ha in year 5 when pruning occurred. In all cases, we used the same value (10%) to quantify the amount of above-ground detritus production that serves as a mineral-soil C input in Roth-C. The size of the IOM pool was estimated at 7.6 Mg C/ha from total soil C (Falloon *et al.* 1998, 2000); the soil clay content used was 12% (K. R. Tate and R. L. Parfitt, unpublished data).

Results

Changes in soil pH and in total and microbial C and N

In L and FH material, pH was lower ($P < 0.01$) in the 400 than 100 stems/ha treatment. Total C and N concentrations in both L and FH material, and microbial C and N in L material, were, however, similar ($P > 0.46$) in both pine treatments (Table 3). Microbial C and N in FH material were, in contrast, lower in the 400 than 100 stems/ha treatment ($P < 0.05$ and 0.001, respectively). On an area basis, total C and N contents in the FH layer were about 2-fold higher in the 400 than 100 stems/ha treatment ($P < 0.01$ and $P < 0.05$, respectively) (Table 4).

In mineral soil, pH values at 0–0.10 m depth were highest in the pasture and lowest in the 400 stems/ha treatment ($P < 0.05$) (Table 3). Soil total C and N ($P < 0.01$) and microbial C and N ($P < 0.01$) concentrations at 0–0.10 m depth were highest in the pasture; only total N differed significantly between the pine treatments, being lower ($P < 0.05$) in the 400 stems/ha treatment (Table 3). The ratios of microbial C/total C and microbial N/total N in 0–0.10 m

depth soil were also highest ($P < 0.05$) in the pasture samples (data not shown). In 0.10–0.20 m depth mineral soil, total C and N and microbial N concentrations were similar in the 3 systems, but microbial C was lowest ($P < 0.05$) in the 400 stems/ha treatment (Table 3). On an area basis, total C and N in the 0–0.10 m layer were highest in the pasture ($P < 0.02$; Table 4); total C and N content did not differ significantly between the 2 pine treatments ($P > 0.10$). For the 0.10–0.20 m layer, total C was also highest in the pasture ($P = 0.03$), but total N did not differ between treatments ($P = 0.10$) (Table 4). For the combined 0–0.20 m soil layer, total C was highest in the pasture ($P = 0.01$). Bulk density did not differ between the 3 treatments at either depth ($P > 0.25$). Soil C content was highest in the 0.20–0.50 m fraction in the pasture ($P < 0.01$), and lowest in the 400 stems/ha treatment; soil N content did not differ between treatments ($P = 0.4$) (Table 4).

C and N cycling rates

In FH material, both CO₂-C production ($P < 0.01$) and net mineral-N production ($P < 0.05$) over 56-day laboratory incubations were lower in the 400 than 100 stems/ha samples (Table 3). *In situ* CO₂-C production was highest in the pasture ($P < 0.01$) and similar in the 2 pine systems (Table 5). In the laboratory, CO₂-C production in 0–0.10 m depth soil was likewise highest in the pasture samples ($P < 0.001$) and similar in the pine samples (Table 3). *In situ* net N mineralisation was also highest in the pasture ($P < 0.01$) and was lower in the 400 than 100 stems/ha treatment ($P < 0.05$) (Table 5). Net mineral-N production in the laboratory was also highest in the pasture ($P < 0.001$) and lower ($P < 0.05$) in the 400 than the 100 stems/ha treatments at 0–0.10 m depth (Table 3). At 0.10–0.20 m depth, both CO₂-C and net

Table 3. Soil pH, total C and N, microbial C and N, and mineral-N concentrations, and CO₂-C (0–56 days) and net mineral-N (0–56 days) production

Values are means with s.d. in parentheses

Sample	Depth (m)	pH	Total C (g/kg)	Total N	Microbial C (mg/kg)	Microbial N	CO ₂ -C (g/kg)	Min-N (mg/kg)	Δ Min-N
<i>Pasture</i>									
Mineral soil	0–0.10	5.6 (0.1)	58 (7)	4.7 (0.4)	993 (108)	190 (17)	1.86 (0.21)	19 (5)	149 (10)
	0.1–0.2	5.5 (0.2)	32 (3)	2.1 (0.1)	196 (21)	35 (4)	0.34 (0.06)	3 (1)	27 (4)
<i>Pine (100 stems/ha)</i>									
Litter	L	4.7 (0)	540 (4)	8.8 (2.8)	9510 (1560)	748 (8)	n.d.	12 (5)	–4 (9)
	FH	4.9 (0.2)	360 (25)	14.0 (1.0)	5790 (960)	1060 (38)	25.6 (1.7)	183 (38)	1250 (163)
Mineral soil	0–0.1	5.2 (0.1)	44 (5)	3.1 (0.5)	526 (105)	74 (10)	0.79 (0.16)	10 (3)	68 (14)
	0.1–0.2	5.4 (0.1)	29 (2)	1.9 (0.3)	163 (21)	32 (4)	0.32 (0.06)	4 (1)	22 (4)
<i>Pine (400 Stems/ha)</i>									
Litter	L	4.5 (0.2)	540 (6)	11.0 (1.0)	9250 (2050)	821 (208)	n.d.	13 (3)	4 (6)
	FH	4.0 (0.1)	350 (28)	13.0 (1.2)	3230 (380)	658 (47)	14.7 (2.1)	105 (39)	908 (113)
Mineral soil	0–0.1	4.9 (0.3)	39 (7)	2.5 (0.6)	340 (104)	45 (9)	0.68 (0.14)	8 (3)	46 (4)
	0.1–0.2	5.2 (0.2)	28 (1)	1.6 (0.4)	121 (24)	24 (13)	0.33 (0.06)	4 (2)	23 (4)

n.d., Not determined.

Table 4. Total C and N pools to 0.50 m depth in the three different systems (<2.0 mm fraction except for 0.20–0.50 m depth increment)
Mean (s.d.) values

Sample	Depth (m)	Bulk density (g/cm ³)	Total C (Mg/ha)	Total N (Mg/ha)
<i>Pasture</i>				
Mineral soil	0–0.1	0.81	46.8 (6.1)	3.8 (0.4)
	0.1–0.2	0.93	29.3 (2.0)	2.0 (0.1)
	0.2–0.5	0.90	53.2 (4.6)	3.3 (0.8)
<i>Pine (100 stems/ha)</i>				
Litter	L	n.d.	1.1	0.01
	FH	0.05	5.7	0.2
Mineral soil	0–0.1	0.83	36.0 (5.8)	2.6 (0.5)
	0.1–0.2	0.89	25.8 (2.3)	1.7 (0.3)
	0.2–0.5	0.85	49.2 (4.7)	3.1 (0.6)
<i>Pine (400 stems/ha)</i>				
Litter	L	n.d.	1.5	0.01
	FH	0.06	10.1	0.4
Mineral soil	0–0.1	0.87	34.3 (7.7)	2.1 (0.6)
	0.1–0.2	0.88	24.6 (2.7)	1.5 (0.4)
	0.2–0.5	0.88	46.4 (3.0)	2.4 (0.6)

n.d., Not determined.

mineral-N production were similar in all treatments ($P = 0.93$ and 0.28 , respectively) (Table 3). After the 56-day incubation, total mineral-N present as nitrate-N averaged 99% at 0–0.10 m depth and 94% at 0.10–0.20 m depth.

Litterfall in plantation forests

Litterfall (both needles and twigs) was higher in the 400 than 100 stems/ha treatment ($P < 0.05$) (Fig. 1), and from 1996 to 1998 averaged about 3.8 Mg C/ha.year and 2.7 Mg C/ha.year, respectively; branches contributed <3% of total litterfall in each treatment. Interannual variation in total litterfall within a treatment differed by <4% across all treatments.

Soil temperature and moisture

Data from the soil temperature loggers, in conjunction with those from the soil respiration measurements, showed soil temperatures were about 2–4°C degree higher in the pasture than the pine treatments (Fig. 2), and generally about 0.5°C higher in the 100 than 400 stems/ha treatment.

Soil moisture did not differ significantly ($P > 0.05$) among treatments in the samples collected for laboratory analyses (data not shown).

Simulation results

Given that we used measured changes in mineral-soil C (0.30 m depth) and soil temperature at Tikitere to parameterise Roth-C, it is not surprising we simulated accurately the change in mineral soil C during the first rotation (26 years) (Fig. 3). While pasture C decayed slowly (Fig. 3) as pasture inputs declined (Table 1), the contribution of 'new' C from *P. radiata* compensated for the loss of pasture C at about 12 years, the approximate time of canopy closure. Most of the decline in soil C occurred during the first 12 years following tree establishment.

As expected, reduced soil temperatures in the forest compared with the pasture (Fig. 2) slowed rates of soil C turnover and reduced the magnitude of mineral-soil C decline during the first rotation of trees. The change in temperature did not, however, alter the general direction of soil C change during the first rotation, although the loss of mineral-soil C was reduced by about 30% when temperature changes were included in the simulations.

Using an identical approach at Puruki to that used at Tikitere, but with a site-specific input scenario, we simulated to within about 8% the measured change in mineral-soil C following conversion of pasture to pine (Fig. 4). Soil C losses were greatest during the first 6 years following tree establishment as a result of the elimination of pasture inputs and the low pine inputs to the mineral soil during this time.

Our results indicate the amount and decay of slash inputs at harvest may influence soil C levels beyond the first rotation. Using our initial estimate of total slash incorporation into the mineral soil (10%) at harvest (7.8 Mg C/ha) (Table 2), and with decomposition spread over 14 years, we found soil C at Tikitere continued to decline after the first rotation; this resulted largely from the lack of inputs from the second-rotation trees during the early stages of growth (Table 1). As a sensitivity test, we then assumed about 70% of the harvest inputs entered the mineral soil (i.e. $7.8 \times 0.7/14 = 3.9$ Mg C/ha.year; a value of 4.0 was used

Table 5. *In situ* soil CO₂ flux, net N mineralisation rates (0–0.1 m depth), and soil temperature (0–0.1 m depth)
Measurements were made in February 1996 and in November 1996; mean (s.d.) values

Ecosystem	Soil CO ₂ flux (g CO ₂ /m ² .h)		Net N mineral-N production (g N/m ² .day)		Soil temperature (°C)	
	Feb. 1996	Nov. 1996	Feb. 1996	Nov. 1996	Feb. 1996	Nov. 1996
Pasture	1.64 (0.49)	2.25 (0.63)	0.072 (0.034)	0.21 (0.02)	18.4 (0.21)	14.5 (0.27)
Pine (100 stems/ha)	0.56 (0.23)	0.35 (0.14)	n.d.	0.053 (0.01)	16.2 (0.18)	10.5 (0.13)
Pine (400 stems/ha)	0.47 (0.16)	0.32 (0.14)	0.011 (0.008)	0.043 (0.008)	14.6 (0.08)	9.4 (0.12)

n.d., Not determined.

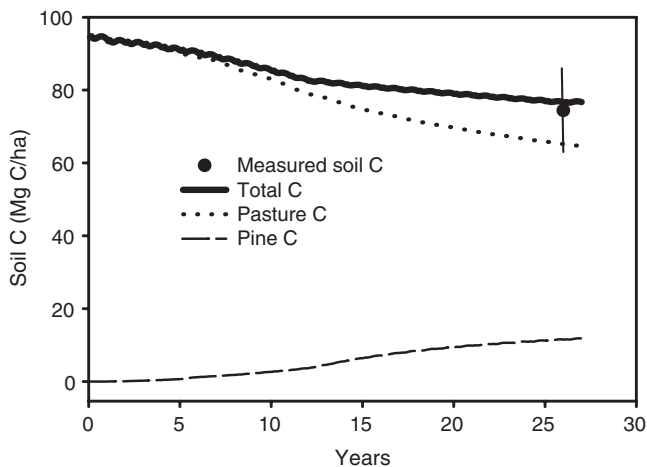


Fig. 3. Simulated changes in pasture C, pine C, and total C to 0.30 m depth during the first rotation of *P. radiata* at Tikitere. Measured soil C data from Scott *et al.* (1999). Error bar = s.d.

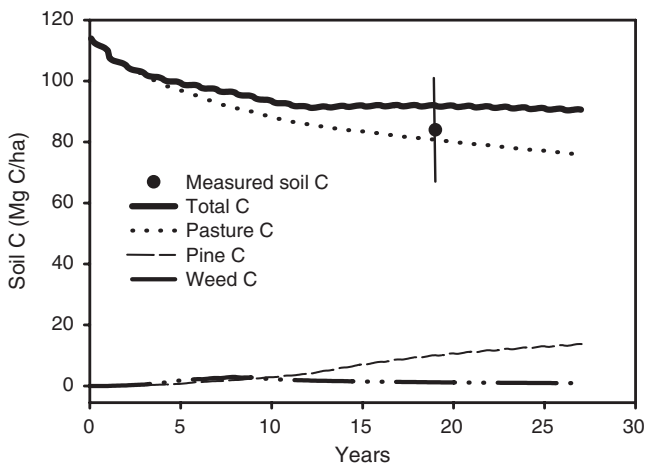


Fig. 4. Simulated changes in soil C to 0.30 m depth during the first rotation of *P. radiata* at Puruki. Measured soil C data from Ross *et al.* (1999) and unpublished data. Error bar = s.d.

in the simulation) were incorporated into the mineral soil each year; even this large total input (about 55 Mg C/ha) still led to a slight decrease in mineral soil C over multiple rotations (Fig. 5).

Discussion

If mineral-soil C inputs remain constant following afforestation, our results showing decreases in soil C and N cycling rates and soil temperature would suggest soil C should increase in the forest system. Decreases in soil temperatures generally reduce soil C turnover rates (Paul and Clark 1989) and increase soil C storage if inputs remain constant. On the contrary, our results indicate significant declines in total (and microbial) C and N, and rates of C and N cycling in mineral soil following afforestation of pasture with

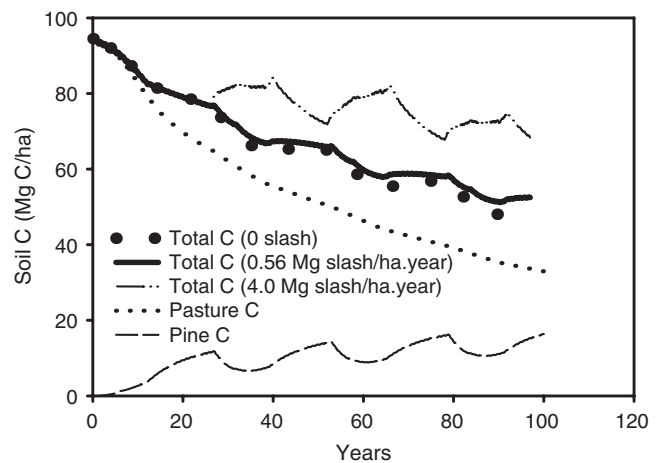


Fig. 5. Simulated changes in mineral soil C (to 0.30 m) over 4 rotations using different levels of slash (0, 7.8, 56 Mg C/ha) and associated annual slash inputs (0, 0.56, or 4.0 Mg C/ha/year; derived by dividing different amounts of slash by 14 years) to the mineral soil. Simulated changes in the relative contribution of the pine/weed C and pasture C pools are shown for inputs of 7.8 Mg C/ha of slash (our estimate of the actual inputs).

P. radiata at Tikitere. This suggests C inputs to the mineral soil changed following afforestation, and these changes influence soil C storage.

Our results indicated a greater decline in mineral-soil C with increased tree stocking rate, as found previously in unreplicated sampling at this site (Saggar *et al.* 2001). While the loss of about 20% of mineral soil C to 0.50 m depth is relatively large, it is not atypical. Multi-temporal soil C measurements in one of the replicates at this site showed small reductions in mineral soil C down to 0.30 m since the time of planting, but almost a 50% reduction in soil C between 0.30 and 0.40 m (Yeates *et al.* 2000). In the same single plot, soil microbiological properties and rates of C and N cycling just before harvest in 1999 were also lower in the plantation forest than in the pasture; it was hypothesised that these differences resulted from (unmeasured) changes in C inputs (Saggar *et al.* 2001). While other studies have shown similar declines in soil C and N cycling rates with afforestation (e.g. Ross *et al.* 1999, 2002), changes in total soil C levels can vary considerably. Based on a comparison of 28 sites, Davis and Condron (2002) found soil C content in the top 0.10 m was about 9.5% lower under plantation forests than pastures. In a larger meta-analysis, soil C was about 10% lower following afforestation of pastures with coniferous species; the changes in soil C were not significant when pastures were afforested with broadleaf species (Guo and Gifford 2002). For the surface horizons, the results from these multi-site syntheses concur with national-scale estimates of soil C decline, based on national soil pedon data, following conversion of pasture to plantation forest in New Zealand (Tate *et al.* 2005).

One possible source of uncertainty in our modelling analysis is the estimate of total below-ground C allocation (TBCA) in the forest. One of the key assumptions associated with using annual litterfall and soil respiration to estimate TBCA is that the forest is at steady-state, such that inputs to the soil equal outputs (i.e. no net change in soil C). Clearly, these plantations are not at steady-state biomass after 26 years (e.g. Maclaren 1996), although rates of C accumulation are decreasing. Previous work (e.g. Gower *et al.* 1997) has suggested that using this steady-state relationship to estimate site-specific TBCA can give erroneous results. More recent work has examined variation in the relationship between litterfall and soil respiration in both 'mature' and aggrading forests (Davidson *et al.* 2002). In aggrading forests, annual soil respiration is about 4 times annual litterfall. In contrast, for mature forests, annual soil respiration is about 3 times annual litterfall. Although the regression equations for mature and aggrading forests differ significantly ($P = 0.04$), the intercepts are similar ($P = 0.38$) and the slopes only slightly different ($P = 0.06$) (Davidson *et al.* 2002). If we increased TBCA by 50%, total belowground C inputs over 1 rotation would increase from about 27.5 Mg C/ha to about 41 Mg C/ha. This increase in inputs, however, is not close to the maximum input used in our harvest scenarios (56 Mg C/ha) (Fig. 5), suggesting that higher TBCA would not change the trends in our results greatly.

If steady-state conditions do not exist, it is still possible to use the relationship between litterfall and soil respiration to estimate annual below-ground C allocation as long as other terms (C pools) are included. For example, Nadelhoffer *et al.* (1998) suggest that in non-steady-state systems TBCA could be estimated using the following equation:

$$\text{TBCA} = R_{\text{soil}} - \text{litterfall C} + \Delta \text{litter C} + \Delta \text{soil C} + \Delta \text{root C} + \text{C export} \quad (4)$$

where $\Delta \text{litter C}$, $\Delta \text{soil C}$ and $\Delta \text{root C}$ represent changes in stocks of forest floor, mineral soil, and roots, respectively, and C export represents C losses through leaching. With afforestation at Tikitere, we are confident that litter C is at or near steady-state based on measurements at several other first- and second-rotation forest sites (e.g. Ross *et al.* 2002; Halliday *et al.* 2003). Mineral soil C is, if anything, declining, which would reduce estimates of TBCA instead of causing it to increase. Little is known about C export (e.g. as dissolved organic C [DOC] loss), but it is unlikely to be large from this soil compared to the other C fluxes. Root biomass is likely to be increasing in conjunction with the increase in biomass. Based on national averages (Maclaren 1996), total ecosystem C storage (including forest floor) increases from about 200 Mg C/ha at 26 years to 205 Mg C/ha at 27 years, giving an accumulation of about 5 Mg C/ha.year. Assuming that 20% of this is root C accumulation (Hall *et al.* 2001), TBCA would increase by about 1 Mg C/ha.year. While this is significant in terms of changing TBCA, the

contribution of these roots to the detritus pool is likely to be small as some of these (larger) roots have relatively slow turnover times.

Simulating the effects of afforestation on soil C storage not only depends on the affect of the steady-state assumption on TBCA and calculation of k_1 (respiration) but also on accurate estimates of the C transfer rate (k_2) between the forest floor and the mineral soil (Halliday *et al.* 2003). This is difficult to quantify, yet it is critical to predicting both land-use change and forest management effects on soil C (e.g. Paul *et al.* 2003; Paul and Polglase 2004). We estimate that about 10% of litterfall C is transferred to the mineral soil each year. One important mechanism that transfers C from the forest floor to the mineral soil is the flux of DOC. In temperate forests of North America, DOC flux from the Oa horizon to the mineral soil ranges from 5 to 24% of annual litterfall (e.g. McDowell and Likens 1988; Qualls *et al.* 1991; Currie *et al.* 1996). This flux is large compared with DOC inputs from precipitation or throughfall, and results largely from microbial activity in the forest floor. While our estimated input of 10% is near the low end of these estimates, these measured DOC fluxes also include some DOC input from precipitation and throughfall (McDowell and Likens 1988; Currie *et al.* 1996). Our annual estimate of C transfer from the forest floor to the mineral soil (0.38 Mg C/ha.year) is within the range of annual DOC flux from the forest floor to the mineral soil across a large number of forested sites (Michalzik *et al.* 2001). Although other processes may transfer C from the forest floor to the mineral soil, the similarity of our estimate with estimates of DOC flux gives us confidence in the value of 10% used in our simulations. Paul and Polglase (2004), point out that simulation of soil C changes using RothC is most sensitive to the partitioning of C during decomposition of debris between that lost as $\text{CO}_2\text{-C}$ and that transferred to soil. We have recently commenced a C-partitioning experiment using ^{14}C -labelled *P. radiata* needles to assess the sensitivity of the model results to these different processes.

Our results suggest soil C may continue to decline through multiple rotations, and that the amount of harvest detritus entering the mineral soil can influence soil C levels. This contrasts with recent literature reviews (Johnson and Curtis 2002; Johnson *et al.* 2002) that suggest little impact of forest management on soil C storage, although the range of responses was large. Previous land-use practices have been shown to influence soil C and N dynamics (e.g. Compton and Boone 2000). New Zealand pastures are highly productive, and their soil C stocks are often higher than those found under native forests (Scott *et al.* 2002). This could predispose these pasture sites to larger soil C losses following afforestation compared to sites previously supporting crops or shrubland vegetation. Other studies also suggest that previous land use is a critical factor that determines soil C changes following afforestation (Paul *et al.* 2002). Comparison of

forest management effects on soil C levels should, therefore, consider prior land use and how it influences initial soil C levels.

To better constrain predicted soil C changes associated with slash production and decay at harvest, we compared predicted soil CO₂ efflux from the Roth-C model with estimates of soil respiration following clearcutting of a *P. radiata* plantation in South Island, New Zealand (Arneth *et al.* 1998). We used similar *k* values to quantify the decay of 5 different C pools (pasture C, litterfall and below-ground *P. radiata* C, weed C, slash [pruning] C, and slash [harvest] C). For the first 6 years following forest harvest, we predicted soil respiration rates ranging from 2.1 to 3.9 Mg CO₂-C/ha.year. This is slightly below the field-based flux estimate (4.0 Mg CO₂-C/ha.year) based on eddy covariance data and chamber-based measurements of soil respiration following clearcutting at the South Island site (Arneth *et al.* 1998). However, our result includes only CO₂-C produced from the decay of slash C that enters the mineral soil, not total C loss from slash decay; this is likely to be substantial, especially in the first few years following harvest. The close agreement between our simulated estimates of soil CO₂ loss and those measured following another clear-cut gives us confidence in our *k* values and estimates of C inputs to the mineral soil.

Other processes could also influence changes in soil C with afforestation. Changes in the availability and distribution of N in an ecosystem can lead to changes in C storage depending on the stoichiometry of different ecosystem components (e.g. vegetation and soil) (Rastetter *et al.* 1992). For example, if large quantities of N are removed from the mineral soil following afforestation and stored in vegetation, a reduction in soil C would result if the C:N ratio remains constant. Sensitivity analysis of simulation results have suggested that the long-term ability of forests to sequester C depends greatly on the ability of soils to retain (immobilise) N (McMurtrie *et al.* 2001). While loss of mineral-soil C with afforestation is not always predictable, loss of soil N is frequently observed, and is generally correlated with soil C losses (Halliday *et al.* 2003). Simulations with the G'DAY model (Comins and McMurtrie 1993), which models N availability and N cycling in forests and pastures, almost always predicts an increase in soil C with afforestation, regardless of changes in the N cycle. One reason for this is that the model appears to over-predict litter C inputs to the mineral soil (Halliday *et al.* 2003). While further work will help refine estimates of the importance of N cycling in forests to C storage, our results, and those of other modelling studies, suggest that changes in C inputs to the mineral soil have a large influence on soil C changes following afforestation.

Conclusions

Our results support previous work showing that afforestation of pastures in New Zealand leads to a decline in soil C

and N cycling rates and soil temperature. It appears likely, however, that changes in C inputs to the mineral soil are the critical factor that determines changes in mineral-soil C storage following afforestation. Whereas many soil C models simulate litter decomposition directly with several parameters, we found, both at Tikitere and at Puruki, that we could simulate mineral-soil C dynamics following afforestation using a single parameter that controls the transfer of litter C into the mineral soil. Our estimate of 10% transfer of litter C into the mineral soil appears to be valid at both sites included in this work. Our results show that management of harvest residues plays a small role in the regulation of soil C changes following afforestation, and highlight the importance of reduced soil C inputs following tree establishment as a time of more rapid soil C changes. While some of the mineral-soil C loss is offset by C accumulation in the forest floor, the sum of mineral-soil C loss and C accumulation in the forest floor at Tikitere still suggests a slight decline in whole-profile C storage following afforestation. It is likely site history/management, leading to differences in initial soil C stocks, is a key factor that determines the magnitude (and direction) of soil C change following afforestation. Most importantly, our results suggest soil C may continue to decline through multiple rotations, and plantation forests may not necessarily be considered 'C neutral' for the purposes of national carbon accounting after 1 rotation.

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