

## Soil carbon storage in plantation forests and pastures: land-use change implications

By NEAL A. SCOTT<sup>1\*</sup>, KEVIN R. TATE<sup>1</sup>, JUSTIN FORD-ROBERTSON<sup>2</sup>, DAVID J. GILTRAP<sup>3</sup>  
AND C. TATTERSALL SMITH<sup>2</sup>, <sup>1</sup>Landcare Research, Private Bag 11052, Palmerston North,  
New Zealand; <sup>2</sup>Forest Research, Sala Street, Rotorua, New Zealand; <sup>3</sup>Small Office Systems, PO Box  
46-024, Lower Hutt, New Zealand

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### ABSTRACT

Afforestation may lead to an accumulation of carbon (C) in vegetation, but little is known about changes in soil C storage with establishment of plantation forests. Plantation forest carbon budget models often omit mineral soil C changes from stand-level C budget calculations, while including forest floor C accumulation, or predict continuous soil C increases over several rotations. We used national soil C databases to quantify differences in soil C content between pasture and exotic pine forest plantations dominated by *P. radiata* (D. Don), and paired site studies to quantify changes in soil C with conversion of pasture to plantation forest in New Zealand. Overall, mineral soil C to 0.10 m was 20–40% lower under pine for all soil types ( $p < 0.01$ ) except soils with high clay activity (HCA), where there was no difference. Similar trends were observed in the 0.1–0.3 m layer. Moreover, mineral soil C to 0.1 m was 17–40% lower under pine than pasture in side-by-side comparisons. The only non-significant difference occurred at a site located on a HCA soil ( $p = 0.08$ ). When averaged across the site studies and the national databases, the difference in soil C between pasture and pine was about  $16 \text{ t C ha}^{-1}$  on non-HCA soils. This is similar to forest floor C averaged across our individual sites (about  $20 \text{ t C ha}^{-1}$ ). The decrease in mineral soil C could result in about a 15% reduction in the average C sequestration potential ( $112 \text{ t C ha}^{-1}$ ) when pasture is converted to exotic plantation forest on non-HCA soils. The relative importance of this change in mineral soil C will likely vary depending on the productivity potential of a site and harvest impacts on the forest floor C pool. Our results emphasize that changes in soil C should be included in any calculations of C sequestration attributed to plantation forestry.

### 1. Introduction

Current research on the global carbon (C) cycle continues to suggest a missing terrestrial sink in the global C budget (Schimel et al., 1995; Melillo et al., 1996b). Globally, changes in land use are a major factor altering the net C balance of the terrestrial biosphere (King et al., 1995; Fernside, 1996; Melillo et al., 1996a). Afforestation is one land-use change

that may explain at least part of the missing C sink and provide a potential short-term mitigation option for decreasing CO<sub>2</sub> emissions (Vitousek, 1991; Moffat, 1997). Several studies have estimated the contribution of afforestation to the global C cycle at both regional (Maclaren and Wakelin, 1991; Turner et al., 1995; Brown, 1996; Shvidenko et al., 1997) and global scales (Nilsson and Schopfhauser, 1995; Brunnert, 1996). Although the potential impacts of vegetation C accumulation during afforestation on global C budgets have been documented, there is little information on changes in soil C following afforestation.

\* Corresponding author.  
e-mail: scottn@landcare.cri.nz

To meet its commitments under the Framework Convention for Climate Change, New Zealand has adopted a CO<sub>2</sub> reductions policy whereby increased C storage by exotic plantation forests (primarily *Pinus radiata* D. Don) offsets increases in CO<sub>2</sub> emissions from energy and industrial sources (MfE, 1997). In 1995, New Zealand's CO<sub>2</sub> emissions from energy and industrial sources were about 7.5 Mt CO<sub>2</sub>-C (MfE 1997; Marland et al., 1998) (0.6 Mt CO<sub>2</sub>-C above 1990 emissions), offset by 3.7 Mt sequestered by exotic plantation forests (MfE, 1997). To date, calculations of C sequestration rates for plantation forests have not included changes in the mineral soil C pool.

Soils are New Zealand's largest terrestrial C reservoir, containing about 2500 Mt C to 0.25 m depth (4260 Mt C to 1 m depth) (Tate et al., 1997). Exotic plantation forests cover about 1.5 million ha, most of which occur in North Island on volcanic soils (Andisols (Soil Survey Staff, 1990)) (Fig. 1). Soils under plantation forests contain about 44 Mt of C in the top 0.1 m (Scott et al., 1997), and about 100 Mt C to 0.25 m (Tate et al., 1997). Soil C in surface mineral soil layers is most likely to respond to changes in land use and management; the top 0.30 m of mineral soil is proposed for national C inventories (IPCC 1996). Given the large reservoir size and potential impact of land-use change, small changes in soil C with afforestation could influence net C accumulation by plantation forests.

Currently, new exotic plantation forest planting often occurs on land previously used for pasture. In 1996, 86% of new forest plantings (84,000 ha) occurred on pastures (both improved and unimproved) (MAF, 1998). Conversion of pasture to plantation forests is accompanied by major changes in soil properties and processes, some (or all) of which could influence the soil C reservoir. First, although total C inputs may not change markedly, organic matter inputs to the soil detrital system differ in quality and distribution (spatial and temporal) (Tate, 1987). In pastures, the majority of C inputs are below-ground (Hart et al., 1990), primarily through root exudation and fine root turnover. These inputs are readily decomposable, leading to high microbial activity in the rhizosphere (Tate et al., 1991). Roots also contribute to total dry matter production in plantation forests (Beets and Whitehead, 1996), but a greater proportion of detrital inputs likely come from

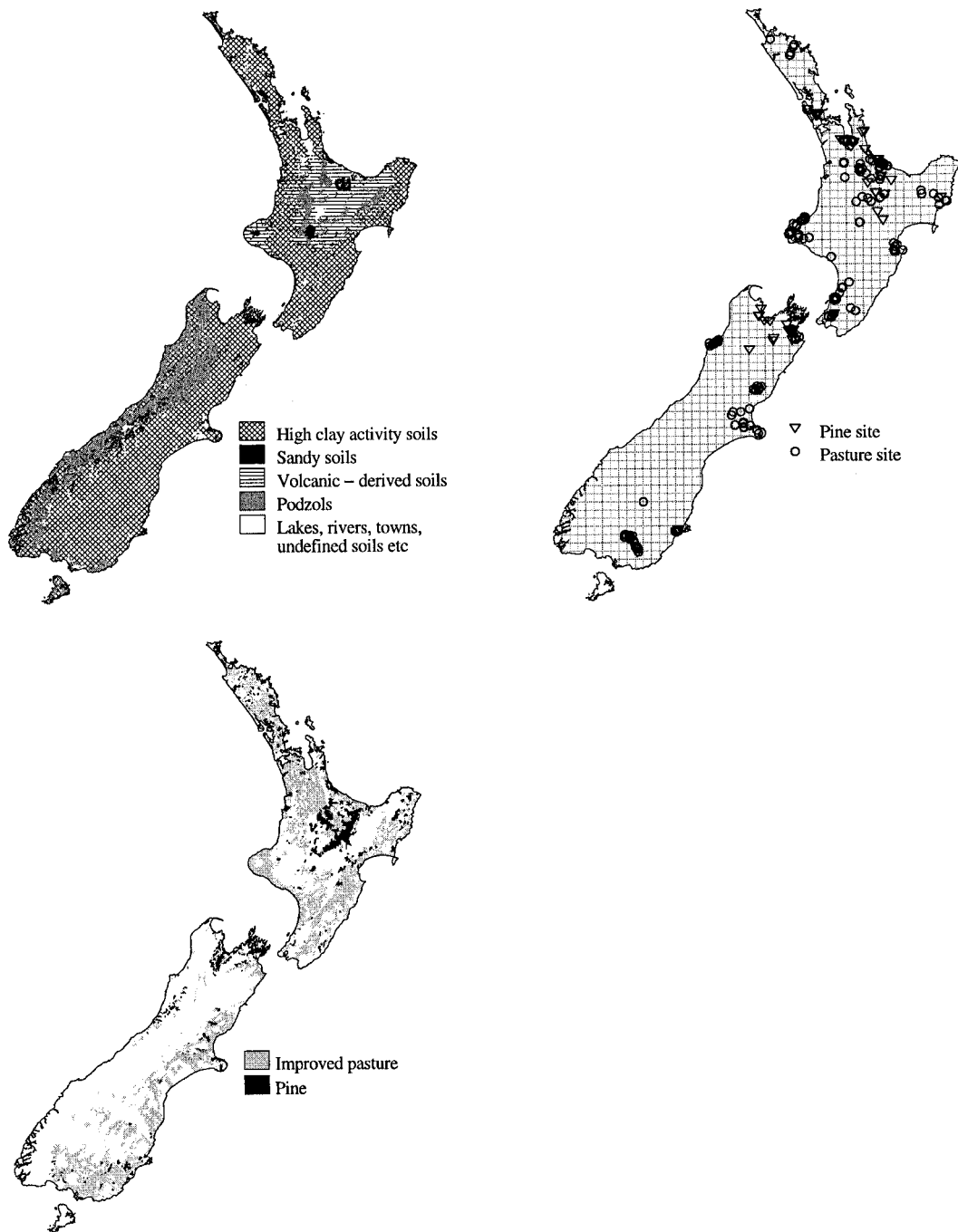
above-ground sources as tree genotypes are selected to maximize above-ground production. This detrital material contains more recalcitrant components (e.g., higher lignin content) than either above- or below-ground inputs in pasture ecosystems (Heal et al., 1997), and is exposed to intermittent drying, which can slow rates of litter decomposition. Second, soil CO<sub>2</sub> production and net N mineralization rates (both in the laboratory and in situ) are generally lower in plantation forest mineral soils (N. A. Scott unpublished data), suggesting decreased soil microbial activity in the forests; low net N mineralization could, however, be caused by rapid immobilization of both nitrate- and ammonium-N (Hart et al., 1994). Third, soil temperatures are generally lower in forest soils due to shading by the forest canopy (N. A. Scott unpublished data). Fourth, soil pH and cation exchange capacity also tend to be lower in plantation forest soils (Davis, 1994; Giddens et al., 1997). Finally, afforestation of former pastures decreases soil invertebrate abundance (e.g., earthworms) (Yeates, 1988). Although it is clear that these soil properties and processes change when pasture is converted to plantation forests, the effect of these changes on total soil C storage is not known.

We therefore examined patterns of soil C content with conversion of pasture to plantation forests. We hypothesized that soil C would increase with afforestation because of decreased soil C and N cycling rates observed in several detailed site studies. We then examined the potential contribution of mineral soil C changes to stand-level estimates of C sequestration by exotic plantation forests.

## 2. Methods

### 2.1. National soil C data

Our first source of data was the New Zealand National Soils Database (McDonald et al., 1988). This database contained information collected over the past several decades, primarily from pasture. We extracted all the soil pit data containing adequate information on mineral soil C concentration, fine-earth bulk density, land use, and geographic coordinates. No methodological screening was possible, but most of the soil C measurements were made by dry combustion (Blakemore et al., 1987). As long as a soil C (or



*Fig. 1.* Distribution of soil (IPCC categories) and land-use types, and the distribution of sample points for pasture and exotic forest soils throughout New Zealand (grid system shown is 20 × 20 km).

Table 1. Site characteristics for pasture/pine comparison of soil C on identical soils

Site	Latitude/Longitude	MAT (°C)	MAPPT (mm)	Stand age (years)	Rotation <sup>1</sup>
Puruki	38°26'S, 176°13'E	10	1500	23	first
Tikitere	38°04'S, 176°22'E	12.7	1490	23	first
Ngaumu	41°03'S, 175°54'E	11.4	1170	24	second
Kaingaroa	38°30'S, 176°31'E	10.7	1480	26	second

Rotation length typically 25 to 30 years.

% organic matter) and bulk density were reported, the data were included in the analysis. The horizon-specific measures of soil C and bulk density were used to calculate mineral soil C content for 0–0.1, 0.1–0.3, and 0.3–1.0 m depth increments.

The second soil C data source was the Forest Nutrition Laboratory soils database held by Forest Research. This database contained data collected from about 1967, primarily from exotic forests throughout New Zealand. Once again, individual records were used only if they contained soil C and fine-earth bulk density information. This database contained some soil C values determined by loss-on-ignition (Nicholson, 1984), where loss-on-ignition was converted to soil C using a factor of 1.724 (Allison, 1965). These two sources of data provided 487 points for New Zealand: 235 in pasture, and 252 for exotic forests containing *P. radiata*.

In both databases, we categorized soils according to the Intergovernmental Panel on Climate Change (IPCC) guidelines (Daly and Wilde, 1997), which are based on the premise that texture and clay mineralogy strongly influence soil C content (IPCC, 1996). These categories include: high clay activity soils (HCA), low clay activity soils (LCA), sandy soils, volcanic soils, aquic soils, organic soils, and podzols (Fig. 1). Podzols are not an IPCC category, but were included because of their large spatial extent and high carbon content.

In most cases, neither database contained information about the temporal dimension of land-use change. The plantation forest soil C database contained some information on previous land use, but no information on pre-plantation soil C levels. Therefore, we did not screen the data for differences in the time an area had been occupied by a particular land use, nor did we screen plantation forest data to include only sites with similar land-use history. We assumed that

the duration of both land-use types was adequate so that soil C content accurately reflected land-use type.

## 2.2. Detailed site studies

We have also investigated changes in soil C with afforestation at four North Island sites (Table 1) containing adjacent areas of pasture and *P. radiata* on the same soil type. Three of the sites are located on volcanic soils (Andisols) and one on a HCA soil. Measurements were also made of forest floor C, information that is currently missing from the databases. At Kaingaroa and Ngaumu, we sampled soils by digging large pits (at least 2 m deep), 5 in pine forest and 5 in an adjacent pasture site on similar soils. Soils in the 0–0.1 m layer were sampled from ground level by compositing 10 cores (50 mm diameter) around each of three pit faces, and subsequent depth intervals were sampled by removing and compositing 10 cores (100 mm diameter, 65 mm long) from each of three pit faces for a total of three samples per pit per depth increment. Forest floor was sampled using a 0.09 m<sup>2</sup> quadrat. At Puruki, soil samples were taken from adjacent pasture and pine sites on similar soils by collecting three independent samples at each site consisting of 25 cores (25 mm diameter) to 0.20 m depth along two transects. The cores were divided into two depths; 0–0.10 m and 0.10–0.20 m. Soil samples for the 0.20–0.50 m layer were taken from a large pit located in, and representative of, the sample area. Forest floor samples were collected with a 0.021 m<sup>2</sup> quadrat. Finally, at Tikitere pasture and pine are treatments in a randomized complete block experiment with three replicates (Hawke 1991). Mineral soil cores (63.5 mm diameter, 20 plot<sup>-1</sup>) were collected along 2 transects through each plot (about 1 ha) to a depth of 0.2 m. Each core was split into

0–0.1 and 0.1–0.2 m depth increments. At alternating sampling locations (total of 10), one core was taken at 0.2–0.5 m depth. All cores taken within a plot were composited for subsequent analyses. We sampled forest floor material at 10 locations in each plot using a 0.1225 m<sup>2</sup> quadrat.

Mineral soils were sieved through a 2 mm sieve, weighed for fine-earth bulk density, then ground for C analysis. Total soil organic C was measured by thermal combustion (LECO FP-2000, LECO Corp., St Joseph, Michigan, USA). Forest-floor material was sieved through a 5.6 mm sieve and analysed for total C by thermal combustion.

Measuring soil C content by depth instead of genetic horizons may bias soil C estimates as a result of soil disturbance during land-use change (Davidson and Ackerman, 1993). Soil C measurement by depth instead of genetic horizon can underestimate changes in soil C storage with cultivation (Davidson and Ackerman, 1993) due largely to re-distribution of C in the soil profile resulting from disturbance and mixing with cultivation. There is no cultivation or major soil disturbance (only animal movement) associated with pasture management. As soil disturbance is minimal and genetic horizons remain largely intact with afforestation in New Zealand, we consider our results based on fixed depths unbiased towards either land use.

### 2.3. Statistical analyses

Data in the two soil databases were analysed for land use (pasture vs. exotic forest) and soil type effects on soil C using analysis of variance (procedure ANOVA, SPSS, 1993). Many of the points in the databases were sampled over a relatively small area (Fig. 1), giving us non-randomized sampling of soil C at the national scale. We examined the effect of non-random sampling by estimating changes in the variance achieved by averaging data points at different spatial scales. First, we averaged data with identical grid coordinates. Then, using a common point as the lower left origin (2000000, 5310000, New Zealand Map Grid Projection), we created three grid systems (5 × 5, 10 × 10, and 20 × 20 km) over the entire country. Soil C values having similar soil type and land use were averaged within each grid cell. The number of unique data points went from 482 to 363 when points with identical grid coordinates

were combined, to 132 when a 10 × 10 km grid was used. New Zealand is covered by about 2500 100 km<sup>2</sup> grid cells, so less than 6% of the grid cells contained soil C data although the land area for pasture and plantation forests represents about 30% of New Zealand's land area (Tate et al., 1997). Hierarchical analysis of variance showed that within cell variance decreased significantly ( $p < 0.05$ ) between each spatial scale, even though the largest change in the number of cells containing data occurred between combining data with identical grid coordinates and data in grids at the 5 × 5 km scale (360 to 175). Because of this significant change in variance, we averaged soil C data for similar land use and soil type within each 10 × 10 km grid cell and used the mean values in our analysis of variance examining the effects of land use and soil type on soil C.

Some of our detailed site studies were pseudo-replicated. However, these sites provide valuable data on forest floor accumulation and a direct comparison of soil C changes with conversion of pasture to plantation forest without confounding effects of factors such as land-use history, soil type, and climate.

### 3. Results

At the national scale, soil C content in the top 0.1 m of mineral soil was 20–40% lower under exotic pine forest than under pasture for all soil types except HCA soils (Table 2). Both soil type and land use had a significant impact on soil C content, and land use interacted significantly with soil type (Table 3). This interaction results primarily from the differential effect of land use on soil C in HCA soils (Table 2). When averaged across the non-HCA soils, soil C was lower under exotic pine forest by about 16 t C ha<sup>-1</sup>.

Averaging soil C data for the same land use/soil type within a 10 × 10 km grid reduced the error df from 469 to 135, but had little influence on the average soil C values for each land use/soil type combination. The only averages that changed significantly at different spatial scales were in pine on sandy soils and podzols. For sandy soils under pine, average soil C increased from 10.2 to 26.4 t ha<sup>-1</sup>, due primarily to the clustering of 24 samples within one 5 × 5 km grid cell. The change was less dramatic for podzols under pine, where

Table 2. Effects of land-use (plantation forest versus pasture) and soil type on mineral soil organic C content for the 0–0.1 m layer.

Soil category <sup>1,2</sup>	Land use	Total C (t ha <sup>-1</sup> )	Sample number (n)	Area <sup>3</sup> km <sup>2</sup>
high clay activity	pasture	41.4 (10.3)	59	37821
	pine	47.0 (12.4)	21	5027
podzols	pasture	62.9 (28.3)	7	929
	pine	48.3 (26.0)	5	1191
sandy	pasture	36.8 (5.7)	8	771
	pine	26.4 (24.1)	4	914
volcanic	pasture	59.9 (15.2)	27	10402
	pine	36.8 (13.9)	12	4841

Averages are based on using a single mean soil C content for each soil type/land use combination within a 10 × 10 km grid cells. Standard deviation shown in parentheses.

<sup>1</sup>IPCC 1996.

<sup>2</sup>Combined area of exotic forest on organic, LCA, and aquatic soils < 1% total exotic forest area.

<sup>3</sup>Total land area of New Zealand about 256000 km<sup>2</sup>.

Table 3. ANOVA table showing the effect of soil type and land-use on soil organic C content (0–0.1 m) using data averaged within 10 × 10 km grid cells

Treatment	F-value	Sig. of p
land-use	9.95 <sub>1,135</sub>	<0.01
soil type	6.04 <sub>3,135</sub>	<0.01
LU × ST	7.94 <sub>3,135</sub>	<0.01

average soil C increased from 38.4 to 48.3 t ha<sup>-1</sup> when the data were averaged inside 10 × 10 km grid cells. For all the remaining soil/land use combinations, estimated soil C content changed by less than 5% as a result of averaging these data within 10 × 10 km grid cells.

Our results from the detailed site studies were consistent with those obtained from the national database analysis. Soil C was 17–40% lower under pine than under pasture in the 0–0.1 m layer, the smallest non-significant difference ( $p > 0.05$ ) occurring at the Ngaumu site on a HCA soil ( $p = 0.08$ ) (Fig. 2). The other three sites were located on volcanic soils, so the lower soil C content observed at these sites was consistent with results from our national assessment (Table 2). Once again, the difference in soil C between the two land-use types was about 16 t C ha<sup>-1</sup> on non-HCA soils.

One of the major differences in soil C between exotic pine forests and pastures results from the

accumulation of forest floor material in forest ecosystems. At the four intensively studied sites, we found that the litter layer C content was similar to, or greater than, the difference in 0–0.1 m mineral soil C content observed for pasture and pine (Fig. 2). Litter C accumulation was lowest at Ngaumu (6.5 t C ha<sup>-1</sup>) (Fig. 2), the site with the lowest fertility status, and highest at Puruki (35 t C ha<sup>-1</sup>). Although C accumulation in litter could offset lower C content in the 0–0.1 m mineral soil layer, it may not offset total profile changes in C due to afforestation. We observed generally lower soil C content under pine than under pasture ( $p < 0.10$ ) at mineral soil depths > 0.1 m at all sites except Kaingaroa (Fig. 2). Preliminary evidence from our databases also suggests lower soil C content in the 0.1–0.3 m mineral soil layer under pine than under pasture, especially for podzols and volcanic soils (40% and 50% lower soil C under pine, respectively). For the whole profile, total soil C storage (including the litter layer) was generally lower under pine than pasture (Fig. 2), but the differences were not significant ( $p > 0.10$ ). At Tikitere, total soil C content to 0.50 m was about 10% lower under pine ( $p = 0.10$ ). At Kaingaroa, mineral soil C to 0.10 m was significantly lower under exotic forest, but total soil C (including forest floor) to 1.0 m depth did not differ significantly between pasture and exotic forest (J. Ford-Robertson personnel communication). This may result from increased soil C variation with depth, reorganization of soil C within

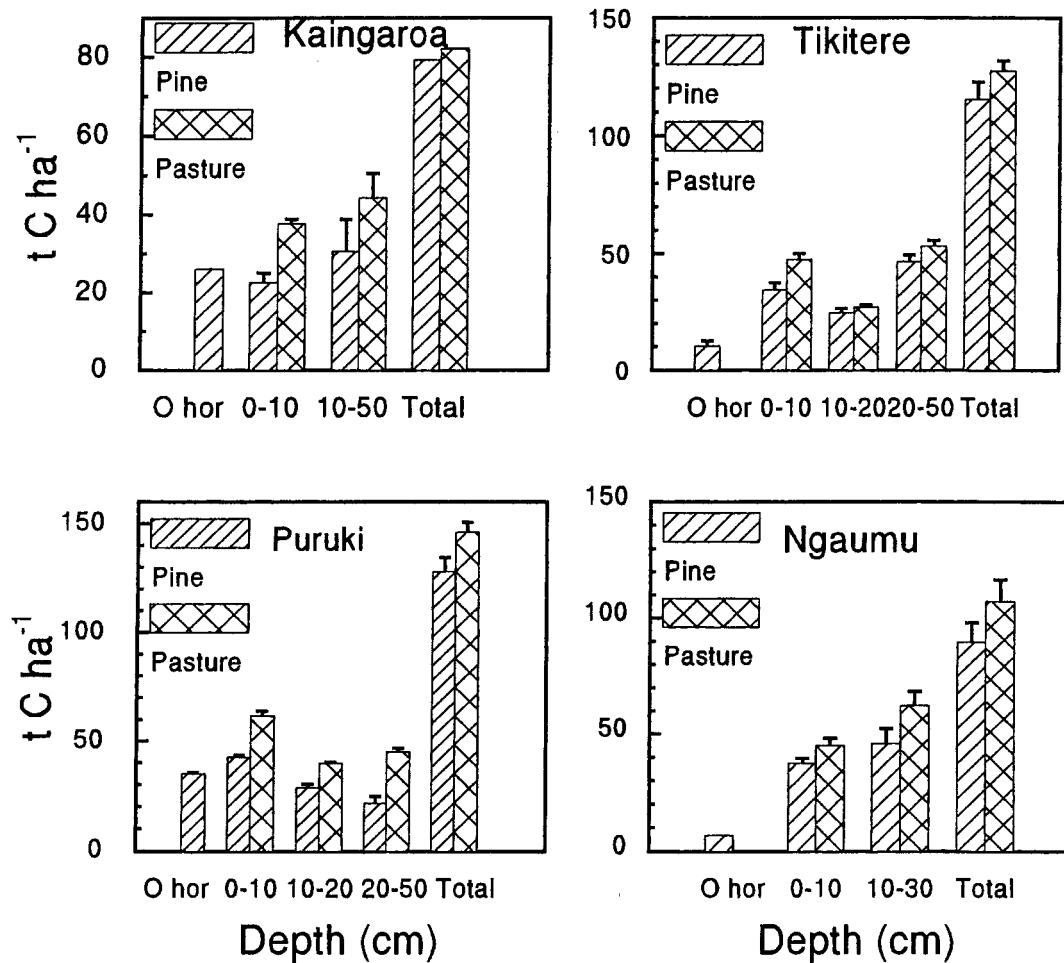


Fig. 2. Soil C content by depth increment from the intensive site studies. Error bar is +1SE; "total" refers to the sum of O horizon and mineral soil C to 50 cm depth (30 cm at Ngaumu).

the profile, or movement of soil C below 1.0 m depth.

#### 4. Discussion

The objective of this work was to examine the implications of converting pasture land to exotic pine plantation forests on soil C content, and whether changes in soil C influence the amount of carbon stored in plantation forests.

Using data collected at one point in time to infer losses or gains resulting from land use can be problematic without knowledge of the temporal

dimension of land use. Because of the lack of temporal data, the results from the national database may not provide accurate estimates of annual changes in soil C resulting from land-use change. However, given the similarity between the results from the national database and our detailed site studies, loss of soil C in surface horizons following establishment of exotic pine plantations on pasture soils seems likely. One alternative explanation could be that soil C increased in pastures instead of decreasing in the plantation forest soils during the period of forest growth. We consider this unlikely for two reasons. First, when pine plantations are planted into pastures, these pastures

have likely been established for over 50 years, so further changes in pasture soil C are unlikely. Second, pastures on a wide range of soil types exhibited no discernable change in soil C content over several decades (Tate et al., 1997).

Land-use history may also influence current patterns of C storage in soils. Previous work has shown that former agricultural practices strongly influence current nitrogen cycling and C storage patterns in North American forests (Aber and Driscoll, 1997; Aber et al., 1997). Whether current plantation forests were established on marginal pasture, highly productive pasture, or scrubland containing a mix of grass and woody species could influence the magnitude of changes in soil C accompanying land-use change. Although we have good information on soil C levels associated with both improved (productive) and unimproved (marginal) pasture, we have little information on soil C levels associated with scrublands. Previous management may also influence current land-use effects on soil C. Future work will quantify soil C levels associated with scrublands and examine historical effects of land-use and management on current spatial patterns of soil C sequestration.

The similar soil C levels under pasture and plantation forests on HCA soils could result from strong interactions between clay minerals and organic matter. These soils contain large quantities of 2:1 clay minerals including mica, smectites, and vermiculite (Hewitt, 1993) that stabilize organic matter (Tate, 1987), and may protect organic matter from losses associated with land-use change. Giddens et al. (1997) reported small or non-significant differences in mineral soil C to 0.1 m depth with afforestation at 10 paired (pasture and pine) sites in the south-central part of North Island, New Zealand. However, their sites were located primarily on HCA soils, where we also observed no significant differences in mineral soil C for the different land uses (Table 2). These results highlight the importance of examining land-use effects on soil C over a range of soil types, as the processes controlling soil C storage with afforestation could vary across soil types and climatic regimes.

#### *4.1. Potential changes in carbon sequestration by plantation forests*

To quantify the potential impact of changes in soil C on plantation forest C accumulation, we

compared changes in mineral soil C with average (national) total forest C accumulation assuming a 30 year rotation ( $112 \text{ t C ha}^{-1}$ ; Maclaren et al., 1995). This forest C estimate includes C in forest floor (coarse and fine detritus) and all above- and below-ground tree components. Current plantation forest carbon sequestration models (e.g., STANDPAK; Hollinger et al., 1993) include C accumulation in the forest floor but do not include any changes in mineral soil C. We examined two possible scenarios for the effect of afforestation on soil C content and C accumulation. The first assumes that changes in mineral soil C with land-use change occur only in the top 0–0.1 m layer (about  $16 \text{ t C ha}^{-1}$  based on the average difference; 0–0.1 m only). This would lead to about a 15% reduction in the C storage potential for one hectare of land converted from pasture to exotic plantation forest on non-HCA soils. In addition to mineral soil C losses, loss of forest floor may occur at harvest (Smith et al., 1994). The second scenario therefore assumes 50% loss of the forest floor layer ( $20 \text{ t C ha}^{-1}$ ) after harvest. If this loss ( $10 \text{ t C ha}^{-1}$ ) is added to the  $16 \text{ t C ha}^{-1}$  loss in the top 0.1 m of mineral soil, the potential decrease in C accumulation would be 23% based on stand level C sequestration of  $112 \text{ t C ha}^{-1}$ . This potential loss of forest floor could also apply to HCA soils, and given the large spatial extent of this soil type could impact strongly on national-scale estimates of C accumulation in plantation forests. These results highlight the need to include changes in both mineral soil and forest floor C as part of the national-scale contribution of exotic plantation forests to the national C budget.

Several factors interact to determine changes in soil C accompanying land-use change. With the conversion of pasture to plantation forest, changes occur in the quality, quantity, timing, and spatial distribution of detrital inputs. Total C inputs to soil may be lower in pine forests as much of the dry matter produced is held in the trees and subsequently removed at harvest. The relatively short rotation length, with associated disturbances and tree removal, could also influence soil C content. Observed soil C changes could result from real soil C losses (e.g., leaching, soil respiration), or reorganization of whole-profile soil C (e.g., movement of mineral soil C into forest floor or deeper in the profile). Whether soils in plantation forests become a long-term source or sink of



C depends greatly on how forest management affects inputs to the soil, particularly during harvesting. Forest harvesting may increase soil C losses through erosion, or increase soil C mineralization due to decreased C inputs after harvest and higher soil temperatures. In contrast, coarse roots remaining after harvest could contribute to the soil C reservoir depending on the long-term fate of this C pool. If much of the forest floor is also lost during harvesting, the potential for these plantation forests to accumulate C could be significantly reduced. As a consequence, a progressive decline in soil C may occur due to the combined effects of short-rotation length and harvest impacts. Further work is required to improve our understanding of the processes causing soil C changes with afforestation, and the influence of factors such as land-use history on these changes. These and other factors are being examined in a

program to establish a monitoring system to quantify land-use change effects on soil C storage.

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