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Land-use change alters New Zealand's terrestrial carbon budget: uncertainties associated with estimates of soil carbon change between 1990–2000

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ABSTRACT

In New Zealand, afforestation and reforestation of grazing land give rise to large (relative to national CO_2 emissions) vegetation carbon (C) sinks. These land-use changes may, however, lead to losses of mineral soil C. Full C accounting may, therefore, require including mineral soil C losses if credits are awarded for vegetation C. To monitor soil C stocks and changes, we developed an IPCC-based soil Carbon Monitoring System (CMS) in which New Zealand is stratified by soil type, climate, land-use and an erosivity index (slope $\times$ precipitation). Georeferenced soil C data were used to assign steady-state soil C stocks to various combinations of these factors (cells). We then used a General Linear Model to compare soil C between cells, and derived land-use effects (LUEs) from this analysis that quantify soil C changes that accompany land-use change. These LUEs were used to predict soil C changes resulting from land-use change between 1990–2000. We tested the CMS by comparing predicted soil C stocks, and changes in these stocks, against more detailed soil C data. Overall, soil C estimates obtained from the CMS are consistent with detailed, stratified soil C measurements at specific sites and over larger regions. However, for grazing-land to exotic-forest conversions, estimates of soil C changes are higher and more variable than those based on paired-site studies. Nationally, soil C losses of 0.9 ± 0.4 Tg C yr^{-1} for all land-use changes over the period 1990–2000 appear likely, with uncertainties arising mainly from estimates of changes in the areas involved, and LUE values for cells with limited soil C data. Changes in soil C in reforested land are likely to be small, but precise area changes, soil C data and detailed paired-site studies are lacking for this key land-use change. By contrast, biomass C accumulation in new exotic plantation forests (afforestation) and native reforestation are 6–9 Tg C yr^{-1}, with C accumulation by afforestation being well quantified. Detailed site studies, process-based modelling and improved area-change estimates are needed to further reduce uncertainties in land-use-change effects on soil C. Our approach could be adapted to countries with country-specific land-use issues different from those in the IPCC default methodology.

1. Introduction

Human intervention has shifted the exchange of C between the land and the atmosphere in favour of the latter, leading to the current increase in atmospheric CO_2 (Keeling et al., 1995). Land-use change has contributed significantly to this increase in CO_2 during both the 1980s and the 1990s (IPCC, 1996). Now, through land-use change and management, human intervention can help reverse this trend in atmospheric CO_2, and this is recognized in the Kyoto Protocol. The Kyoto Protocol paves the way for countries to use terrestrial C sinks to help offset increased CO_2 emissions (UNFCC, 1998). If Annexe One countries ratify the Protocol, credible national strategies for reporting

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and reducing CO_2 emissions will be needed, based on internationally recognized methodologies and subject to periodic auditing. Reduction in uncertainty in the global C cycle should also result from reduced uncertainties in national-scale C budgets through the use of well coordinated national databases, monitoring networks and historical knowledge (Tian et al., 1999). Although soil C is not explicitly included in the current version of the Kyoto Protocol, the IPC has provided guidelines to help countries include soils in their national C budgets (SRLUCF, 2000). These guidelines are based on land-use changes primarily related to agricultural soils, but these particular changes may not be of equal importance to all countries.

In New Zealand, conversion of grazing land to both native shrublands and to exotic plantation forests has been the major land-use change over the past 15 years. Both of these land-use changes have resulted in large increases in vegetation C uptake and storage. Although stocks and harvest-removals are accurately quantified for exotic forests (Marshall et al., 2001), vegetation-C accumulation in shrublands by natural regeneration is relatively poorly understood (Scott et al., 2000; Trotter et al., 2001; Coomes et al., 2002). A further uncertainty in New Zealand's national terrestrial C budget is the effect these land-use changes have on soil C (Tate et al., 2000). To reduce this uncertainty, we have been developing a national Carbon Monitoring System (CMS) for soil to quantify baseline C stocks (Scott et al., 2002) and changes associated with land-use change. We here test the hypothesis that, nationally, recent land-use changes have resulted in an overall loss of soil C. Our approach was to refine the national CMS from previous work (Scott et al., 2002), test the system against field data at different spatial scales, and then apply it to estimate the effects of key land-use changes on New Zealand's soil C stocks in the period 1990–2000.

2. Methods

2.1. Description and refinements of soil CMS

Our national soil CMS (Scott et al., 2002) uses national soil pedon data to derive country-specific estimates of soil C stocks for combinations of seven soil categories, two climate classes and ten land-cover classes. This same approach can be used to estimate soil C changes that accompany land-cover change (Fig. 1).

This IPCC-based default methodology (IPCC, 1996) has been modified from our previous work (Scott et al., 2002) by (a) adding an erosivity index to soil type, climate and land cover/use as the major drivers of change in soil C (Fig. 1) and (b) refining the soil C database. The erosivity index was calculated as the product of mean annual precipitation and slope. It indicates the propensity of a site to erode, and is a useful additional predictor of soil C (Giltrap et al., 2001). A 25-m resolution digital elevation model (DEM) was used to compute the national distribution of slope, aspect and other topoclimatic variables.

Due to imprecise grid references, slope information for the points in the geo-referenced soil C database was derived from field survey data rather than from the DEM. Therefore, soil C data without slope data are omitted from the statistical analysis (see section 2.2) as they lack values for the erosivity index. This omission is acceptable at the level of the national CMS (308 out of 400 points retained for 0.3 m), but unacceptable for the important exotic forest land-cover class (5 out of 18 points retained for 0.3 m). Therefore, the erosivity index has been excluded from analyses presented here involving exotic forest. Additional slope data and/or additional soil C and slope data must be acquired for exotic forests for future work involving the erosivity index.

The CMS is based on the assumption that soil C values contained in the national soil pedon database represent equilibrium soil C values for each combination of soil/climate/land-cover and erosivity index (where appropriate). Any soil pedon data identified from disturbed sites at the time of sampling were not included. Furthermore, it is assumed that changes in land cover or land use are the key drivers of annual and decadal changes in soil C. The other drivers (soil/climate/erosivity index) are assumed to be constant at this temporal scale. Estimating changes in soil C relies on periodic national land-cover updates (e.g. MAF, 2001). Retrospective analyses of land-cover-change effects on soil C are also possible for some land-use changes (e.g. exotic forests), but for others (e.g. regenerating indigenous shrublands) area changes are poorly quantified, and soil C data are limited.

2.2. Modification to the soil C database.

We reduced the number of soil C values used previously (Scott et al., 2002) by eliminating sites where land-use change had occurred less than 10 yr

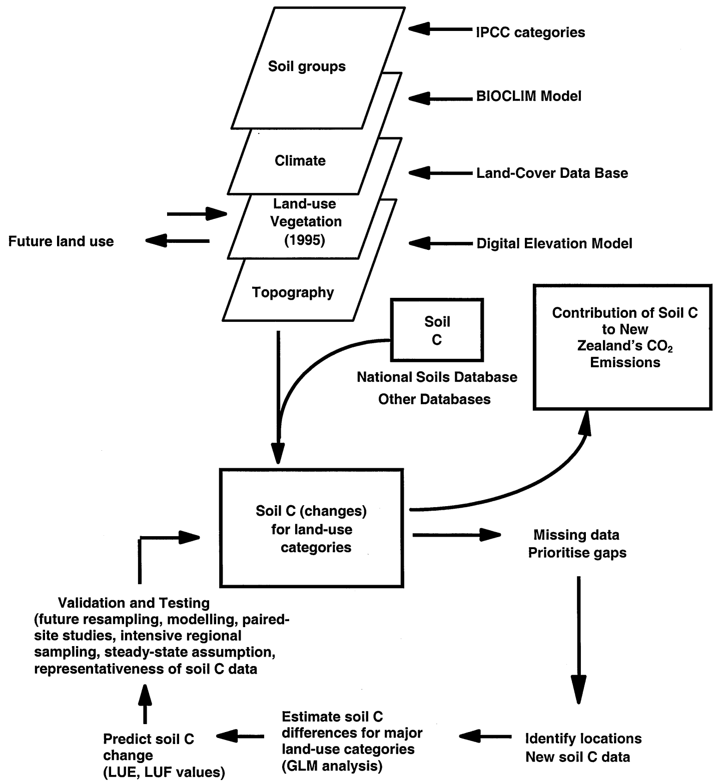

Fig. 1. Schematic of the development and testing of the soil Carbon Monitoring System.

before sampling. Then we examined the validity of including individual pedon values in close proximity to one another. Previously (Scott et al., 2002) we eliminated duplicate sites with identical grid references, soil type and land-cover, retaining only the first site from each cluster. With this further database refinement, we combined clusters of data points with grid references <1 km apart to give single composite points representing the mean value of all the nearby points. Estimates of stone content were also revised by taking account of stone mass recorded in both the field and laboratory records.

2.3. *Quantifying land-use effects on soil C and associated uncertainties*

We used a Generalized Linear Model (GLM) (Giltrap et al., 2001) to examine the impact of the various factors on soil C, based on soil C values for 448 sites (0–0.1 m depth). Fewer sites had soil C data for 0.1–0.3 m (400) and 0.3–1.0 m (270). Major factors tested for the model included combined soil/climate categories, updated land-cover categories, the interaction of soil/climate and land-cover categories, and seven other numeric predictors derived from topography and climate. Stepwise regression was used to find the best predictive model for both untransformed and log-transformed soil C data. The land-use effects (LUE) for the untransformed data (additive model) were taken as measures of the expected equilibrium change in soil C following land-use change. The corresponding effects for the log-transformed data (multiplicative model) were transformed to land-use factors (LUF). The corrected R^2 values for the two models were compared to see which of the additive or multiplicative models better described the effects of land cover on soil C.

LUE and LUF values were expressed relative to the grazing land category, as this category was most commonly involved in land-cover change. This means that grazing land is automatically assigned an LUE of 0 and an LUF of 1. The standard errors were estimated directly from the GLM analysis, and apply to the comparison of grazing land with any other category. National soil C inventory values (and their standard errors) were also estimated from the GLM using the areas of the soil/climate and land-cover categories and the mean values of the other predictor variables estimated from digital soil, climate, land-cover and topographic surfaces using a Geographic Information System (GIS).

2.4. *Testing predictions of soil C stocks using the soil CMS*

In a previous test of the CMS, Scott et al. (2002) compared predicted soil C stocks against measured soil C concentrations and bulk density in a region with unimproved grazing lands. Here, we have undertaken further testing of the CMS, including comparison of (a) predicted (CMS) and measured (unbiased field sampling) soil C values for Temperate Volcanic soils under grazing land, (b) soil C stocks at the landscape (ca. 6000 ha) scale with detailed, stratified sampling within this region, and (c) soil C changes following conversion of improved ryegrass/white clover grazing land to *Pinus radiata* D. Don plantations using the soil CMS and independent paired-site studies. We also examined the steady-state assumption and effects of management changes in grazing lands on soil C (14.0 M ha) using published and unpublished site studies (Saggar et al., 2001).

2.4.1. Comparison of soil C stocks using soil C from historical (non-random) and current random sampling. We tested the representativeness of soil C values for Temperate Volcanic soils under grazing land (improved pasture) by comparing values in the CMS to current values obtained from random field sampling. We selected this "cell" because it is widespread, the area was accessible for sampling, and it has a well established mean and variance (from non-random sampling of modal soils; $n = 29$ for 0–0.3 m depth).

To select random sampling locations, we laid a 5 km $\times$5 km grid over the entire national area of this cell. Grid squares containing 30% or more of the Temperate Volcanic soils under grazing land were numbered, and 20 grid squares were selected randomly for sampling. The centre of the grid square was designated as the sampling point for each of the twenty sites. If map data indicated that the centre of the plot was not the correct soil or land-cover type, the sample point was shifted 2.5 km north (to the edge of the grid square). If the second point was rejected, the next point was located 2.5 km to the east and so on around the compass until an appropriate sample site was found, otherwise that grid square was rejected and the next randomly selected grid square selected instead. Once in the field, if the grid centre was located in either the wrong vegetation or soil category, or if the proposed sample point lay on a road, building or was occupied by a water body, it was rejected. This occurred for six of the twenty sites. An alternative sample point was then selected by moving a random distance (100–500 m) north of the original point. If the second point was rejected, then another was randomly selected east of the original point, and so on around the compass until an alternative sample point could be established within the grid square. Sampling points were initially located on a 1:250 000 NZMS 262 topographical map, and then transferred to a 1:50 000 NZMS 269 topographical map for more accurate location in the field.

Soil sampling involved laying out a 20 m$\times$20 m plot and dividing it into four quarters. A point was randomly located in each quarter, and eight 25 mm diameter cores were taken to 0.3 m depth at 0.25 m intervals on either side of the random point. Soil bulk density was determined using single 98 mm diameter $\times$100 mm deep cores to 0.3 m depth. Soil profiles were described at each of the four random points.

2.4.2. Landscape-scale prediction of soil C stocks. Measured soil C values were obtained for the 0–0.1 m layer from 204 sites in North Canterbury (Jake Overton, personal communication). The North Canterbury sites were selected to represent an area of about 6000 ha in the central part of South Island. At each site, field teams sampled 0–0.1 m soil at three designated locations in each transect plot using a 0.1 m core (98 mm diameter). If a site could not be sampled due to rocks, it was considered 100% rock. Land-cover type was recorded by the field team, and was used in conjunction with the associated soil/climate category to calculate a predicted value for the 0–0.1 m soil C content at these locations using a GLM that did not include the slope $\times$ precipitation factor. This factor was excluded from the model because it affected soil C in the relatively low rainfall North Canterbury region by less than 1 Mg C ha^{-1}. Recorded land-cover categories generally matched categories defined in 1996 for the Land Cover Database (LCDB) (MAF, 2001) and in 1985 for the VCM (Newsome, 1987). The exception to this was for shrubland. No shrubland sites were sampled according to the field classification, but the LCDB (MAF, 2001) indicates large areas of shrublands in the sample area. To overcome this problem, field plots recorded as unimproved grazing land but categorized by the LCDB as shrubland were recorded as shrubland. The CMS predicted soil C value was then compared with the measured C value for each of these reclassified sites to assess the ability of the soil CMS to predict soil C stocks in shrublands within this region.

2.4.3. Comparison of CMS with paired-site results. Data from paired sites comparing soil C values directly where grazing land was converted to planted exotic forests were assembled from published papers, student theses and some unpublished studies (Tate et al., 2001; Davis and Condron, 2002). The forest sites all contained plantings of *P. radiata*. Paired site studies where the forest was not planted into grassland or where there was some uncertainty have been excluded. Separate analyses were performed for 0–0.1 m and deeper depths. Paired sites included forests >10 yr in age for the 0–0.1 m depth and ≥ 7 yr for deeper depths.

A total of 28 paired sites had information on both soil C concentration and bulk density to 0.1 m depth for exotic forests >10 yr in age: 17 sites contained forests 10–20 yr old, and 11 had forests more than 20 yr old. A further 17 sites had C concentration data only available. Soil C in grassland and forest soils was compared statistically using the paired *t*-test. Where sites had been sampled more than once, a mean value was calculated and used for the *t*-test if the difference between sampling times did not exceed 3 yr. Where the difference was more than 3 yr, only the most recently collected data were used. One exception was the Tikitere site (which had been sampled on three occasions). Data for this site from Scott et al. (1999) were used in preference to those of Yeates et al. (2000), because the data were based on measurements from three replicates, rather than from one replicate. A smaller number (7) of paired-site results were available for deeper soil depths (ranging from 0.2 to 0.5 m) from published and unpublished data. These data were compared with predictions from the CMS for the 0–0.3 m depth increment.

2.4.4. Soil C in grazing lands. One critical assumption in the soil CMS is that soil C values in the national database represent steady-state values for a given "cell." Most of the current land-use changes in New Zealand involve grazing land, so it is particularly important to test the steady-state assumption for this land-cover type. The steady-state assumption was tested in established grazing lands using three different approaches (Saggar et al., 2001): (i) published soil C data from long-term research sites, (ii) comparison of soil C levels between archived (30–50 yr) soils and soils subsequently resampled from the same sites (Tate et al., 1997), and (iii) comparison of soil C levels at experimental sites examining different fertiliser management regimes.

2.5. Areal changes in land-cover since 1990

Currently, New Zealand lacks a comprehensive database that provides continuous geo-referenced information over the period 1990–2000. We derived estimates of land-cover areas and areas of change using various national agricultural statistics collected since 1990 (MAF, 2001), from the VCM (Newsome, 1987) and from the LCDB (MAF, 2001). Uncertainty in these areal estimates of land-cover change between 1990–2000 arise from: (i) limited systematic data in agricultural statistics for areal extent of improved and unimproved (including tussock) grasslands and other land-cover types; (ii) inconsistencies in areas under similar land cover as determined by the LCDB and the VCM, e.g. spatial correspondence between the VCM and LCDB for shrublands is poor; (iii) recognition of cropland in the VCM but its inclusion in grazing land in the LCDB; (iv) reclassification of farm land to non-agricultural land in agricultural statistics published between 1986 and 1990; (v) definitional issues between VCM and LCDB due to diffuse boundaries between grasslands and shrubland, and shrubland and forest.

Because of these difficulties, we compared and reexamined national statistical data sets on land-cover to approximate current national figures (year 2000) for each major land-cover that corresponded more closely with the LCDB than the VCM or other agricultural databases (MAF, 2001), and then determined the areal changes under each land-cover since 1990.

3. Results and discussion

Here we present new results based on refinements of the previous CMS (Scott et al., 2002), use the system to quantify (with uncertainty) changes in soil C associated with land-use change, test the systems against field data at different spatial scales, and then apply the system to quantify changes in soil C for the period 1990–2000 for the major land-cover types (Table 1).

Table 1. *Areal extent of major land-cover types in New Zealand in 2000*

Land-cover	Area (Mha)	Area (% of total land)
Grassland	13.61	50.9
Forest land		29.9
Indigenous forest	6.25	
Planted forest	1.73	
Cropland		1.2
Arable and grain	0.21	
Horticulture	0.09	
Shrubland	2.65	9.9
Others	2.20	8.2
Total land	**26.7**	**100**

Data Sources: Ministry of Agriculture and Forestry (MAF) Statistics New Zealand Agricultural Production Survey (2001); Land cover database 1996/97 satellite imagery; National Exotic Forest Description (2000), Tate et al. (1997).

3.1. Modification of soil C database

The main impact of combining all points within 1 km of one another was to reduce the number of points in the Exotic Forest category from 113 to 29, 56 to 18 and 7 to 4 for 0–0.1, 0.1–0.3 and 0.3–1 m depths, respectively. The overall effect of these modifications on the nationally averaged soil C estimates for all soil depths was not significant (data not shown).

3.2. GLM results and national soil C stocks

The GLM showed that the additive model using soil/climate + land cover + (slope × precipitation) gave the best prediction of 0–0.1 or 0–0.3 m soil C. The additional variation explained by adding the soil/climate × land cover interaction, and all other predictor variables, was not statistically significant ($P = 0.43$ for 0–0.1 m; $P = 0.881$ for 0–0.3 m soil C). The adjusted R^2 values from the additive model were higher than those for the corresponding multiplicative model for both 0–0.1 m (0.37 vs. 0.28) and 0–0.3 m (0.44 vs. 0.37) depths.

The most important consequence of these results is that a single land-use effect (LUE; Table 2) can be applied across all soil/climate categories and (slope × precipitation) values. The higher R^2 values for the additive model compared to the multiplicative model indicate that it is more appropriate to use additive LUEs rather than land-use factors (LUFs; Table 2). If required, however, LUEs may be converted to LUFs by the equation

$$\mathrm{LUF} = 1 + \mathrm{LUE}/C_0$$

where C_0 is the mean value for grazing land. For 0–0.3 m soil C, the estimate of C_0 was 114 ± 7 Mg ha^{-1}.

The GLM analysis compensates for confounding responses from other factors included in the model arising from land-use selection effects. For example, sites with high precipitation × slope occur mainly in mountainous terrain, and these sites are much more likely to be covered by indigenous forest than grazing land. Precipitation × slope is negatively correlated with soil C and, if excluded from the model, the soil C for indigenous forest is likely to be overestimated compared with grazing land. If a precipitation × slope term is included, however, this confounding effect is corrected. The GLM model also indicates that no significant confounding effect should be expected for the other variables considered. A comparison of the selected model with other GLM models (e.g. land cover alone, land cover + soil/climate, land cover + soil/climate + all numeric predictors) indicated the LUEs did not vary greatly between models (one exception was for indigenous forest LUE, which changed from −34.4 to −49.0 Mg ha^{-1} when the precipitation × slope term was omitted). The factors considered include those (climate, topography and soil) most likely to influence

Table 2. *Land-use effects (LUE) and land-use factors (LUF) for 0–0.3 m soil C derived from GLM analysis (relative to grazing land)*

	VCM-based[a]		LCDB-based[b]	
Land-use category	LUE (Mg ha^{-1})	LUF	LUE (Mg ha^{-1})	LUF
Cropping land	−14.5 ± 10.2	0.87	n.d.	n.d.
Planted forest	−51.4 ± 18.6	0.55	−51.1 ± 18.6	0.55
Shrubland	−19.3 ± 7.7	0.83	−19.3 ± 7.6	0.83

±indicates standard error; n.d., not determined.
[a]Newsome 1987; [b]MAF, 2001.

soil C. Therefore, it seems reasonable to conclude that confounding effects of other environmental variables not considered (or non-linear responses to those considered) are unlikely to lead to substantial systematic errors in the estimated LUEs.

We constructed national estimates for soil C storage using the LUEs with a national mean value for precipitation × slope (39 m degree). This model yielded national soil C to 0–0.1, 0.1–0.3 and 0.3–1 m depths of 1130 ± 40, 1300 ± 60 and 1300 ± 160 Tg, respectively. Omitting precipitation × slope gave similar estimates (1140 ± 40, 1320 ± 60 and 1330 ± 160 Tg, respectively). These estimates agree closely with earlier estimates (Scott et al., 2002) for 0–0.1 m depth, but are lower by 8% for 0.1–0.3 m and by almost 20% for 0.3–1 m depth.

While development of the CMS is well suited to deriving and revising national estimates of soil C storage, it must also accurately predict soil C storage in each land-cover category (or cell) to best describe soil C changes resulting from land-use change. We emphasize that the current structure of the CMS is almost ready for use as an operational system. However, major data gaps still exist for some cells, and need to be filled before the system can be used most effectively. Examples of these data gaps for key land-use changes are discussed in subsequent sections. In addition, lack of soil C data below 0.3 m depth remains a considerable source of uncertainty. Another difficulty involves Humid Temperate Podzols, which represent a relatively large area nationally and are well represented in the CMS database, even for 0.3–1 m. Four soils in this soil/climate category have 0.3–1 m soil C over 300 Mg ha^{-1}, while the mean for the category is 88 Mg ha^{-1} ($n = 27$). These unusual data appear to 'destabilize' the statistical analysis for 0.3–1 and 0–1 m depth increments. Surprisingly high values result from the GLM for soil C in Humid Temperate Podzols, while strong negative LUEs result for land-cover categories that did not include any of the unusually high soil C samples. For the use of the CMS as an operational tool, difficulties such as this will be best corrected by increasing sample numbers in the highly variable soil categories. Substitute methods, such as paired-site analyses and process-based models, may also be required to reduce uncertainty.

3.3. *Testing the CMS at different scales*

3.3.1. Testing the representativeness of national soil pedon data. As part of ongoing testing of the CMS, we compared soil C values in the CMS database with values obtained from random field sampling in the Temperate Volcanic soil under grazing land (improved pasture) (Table 3). Mean values for soil C mass determined by the two methods were similar, the randomly derived values being within 5.2%, 1.6% and 1.4% of the database values for the 0–0.1, 0.1–0.3 and 0–0.3 m layers, respectively. Means derived from random sampling for the three depths were well within the 95% confidence limits of the database values. With one exception (bulk density of 0.1–0.3 m layer), the random means for bulk density and C concentrations were also within the 95% confidence limits of the database values. These results indicate that the soil C concentrations of the historical samples represented in the CMS are representative of current soil C concentrations for this cell. They also suggest that there have been no discernible land-management effects on soil C concentrations, since database samples were originally collected.

Table 3. *Mean bulk density (BD), C concentration and C mass values for the Temperate Volcanic soils under grazing land determined by the soil CMS and by random sampling*

		Mean	
Soil depth (m)	Attribute	Database	Random
0–0.1	BD (Mg m^{-3})	0.81 ± 0.03	0.81 ± 0.03
	C (%)	7.79 ± 0.53	7.73 ± 0.52
	C (Mg ha^{-1})	60.0 ± 2.8	63.1 ± 5.3
0.1–0.3	BD (Mg m^{-3})	0.81 ± 0.04	0.90 ± 0.05
	C (%)	4.69 ± 0.40	3.92 ± 0.47
	C (Mg ha^{-1})	71.50 ± 5.3	70.3 ± 8.7
0–0.3	C (Mg ha^{-1})	131.5 ± 7.7	133.4 ± 13.8

±indicate standard errors with $n = 29$ for database and 11 for random sampling.

3.3.2. Regional test of the CMS in South Island. Previous regional-scale (24 000 ha) testing of the CMS (Scott et al., 2002) showed that including soil C variation due to both soil and climate in the system produced better agreement with results from detailed, stratified soil sampling than did soil class alone to a depth of 0.2 m. Here, we report results from a comparison of the CMS with stratified soil C measurements in an area of South Island (about 6000 ha), containing a wider range of land-cover and soil/climate categories than were present in the previous test (Fig. 2).

Results from this comparison suggest that the national CMS systematically overpredicts the mean

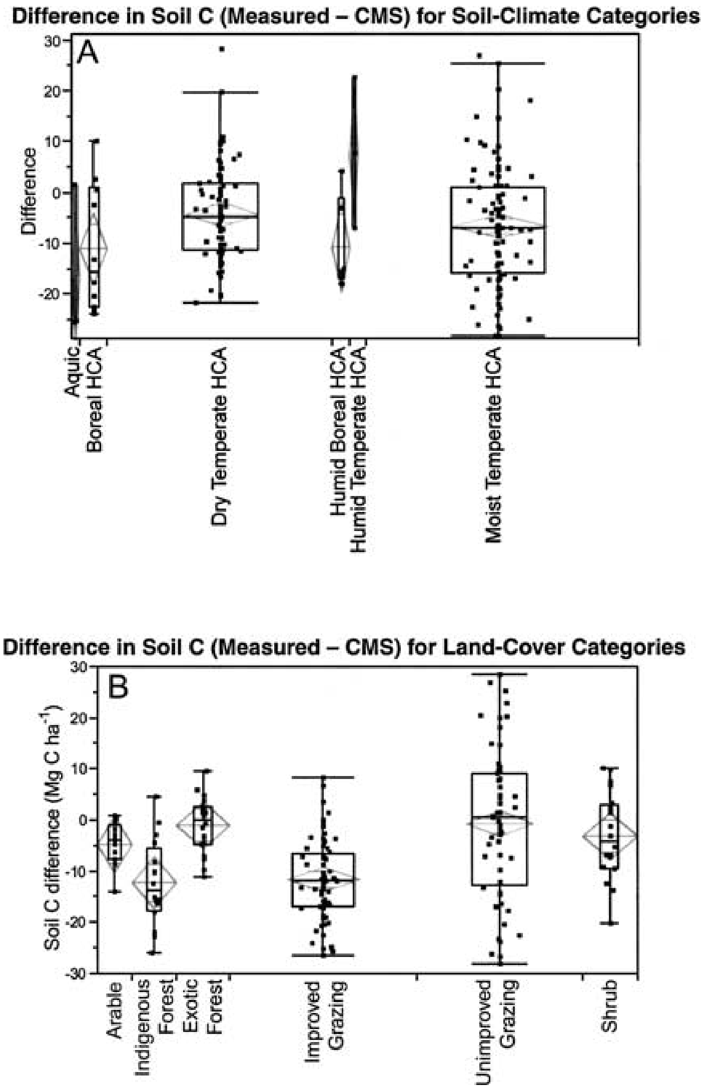

Fig. 2. Differences between soil CMS predictions and 0–0.1 m soil C measurements for the North Canterbury transect area for: (A) soil climate categories, (B) land-cover categories. Box plots indicate quantiles and outliers. Diamonds indicate means and 95% confidence intervals. Width of categories and box plots are proportional to the number of samples in the category. HCA, high clay activity.

soil C for some soil/climate and land-use categories (Figs. 2A and 2B) in this region. The regional means are not significantly different from the CMS prediction for shrublands, planted forests and unimproved grazing land, but are significantly lower for improved grazing land, broadleaf (indigenous) forests and, to a lesser extent, arable land.

Stoniness and slope may account for some of the differences between measured local mean soil C concentrations and the corresponding national CMS predictions. For the land cover that differs most from the CMS predictions (improved grazing land), over 90% of sites contain no stones in the national CMS database (mean = 19 Mg stones ha^{-1}). By contrast, 65% of the sites sampled in improved grazing lands in the transect area have over 20 Mg stones ha^{-1}. The mean stone content for the national CMS database and the current field sampling are 19 and 265 Mg stones ha^{-1}, respectively. In contrast, exotic forests in the sampled area are stonier but less steep than those in the CMS, and compare favourably with soil C values predicted by the CMS. Despite this, regression analyses comparing stoniness or slope to soil C in the sampled area suggest a slight negative relationship ($0.04 < P < 0.10$); however, these marginally significant relationships fall an order of magnitude short of accounting for the observed differences between measured and predicted soil C.

The existence of systematic differences between the local mean soil C values and the national CMS suggest it may be possible to improve the CMS further by identifying other critical, mapable soil properties or developing process-based models that account for these differences. On the other hand, substantial random scatter within soil and land-cover categories (Figs. 2A and 2B) indicate a substantial part of the unexplained variance in soil C is caused by factors that vary even within this limited area. Despite the unexplained local effects, the CMS performed relatively well in this region, given the following provisos. First, for land-use categories altered by human management (improved grazing land and arable land), the large differences between measurements and the CMS are not representative of these categories nationally. Second, soil C stocks in the land-cover categories most likely to undergo change (exotic forests, unimproved grazing land and shrublands) were predicted accurately by the CMS.

3.3.3. Testing the CMS against detailed land-use comparisons. In addition to testing the CMS at larger spatial scales, we also examined how accurately the CMS quantifies changes in mineral soil C compared with results from several paired-site studies where exotic forest has been established on grazing land (Tate et al., 2001; Davis and Condron, 2002). The CMS predicts larger soil C differences (and therefore LUEs) than indicated by results from paired-site sampling (Fig. 3). The discrepancy between the two methods may result from (a) the practice before about 1990 of planting forests on land considered unsuitable for pastoral farming, leading to a bias in the CMS based on hitorical site selection for exotic forests (i.e. sites with potentially low soil C values); (b) the influence of a small number of points in the CMS for exotic forests with very low C values (e.g. <10 Mg C ha^{-1} 0–0.1 m); and (c) the paucity of 0–0.3 m data for *P. radiata* sites. Given these possible problems with the data in the CMS, we decided to use a limited number (7) of paired-site results for the national calculation (Table 4), as they are more appropriate for post-1990 forests. Future work will overcome this potential bias by additional sampling of recently established exotic forests.

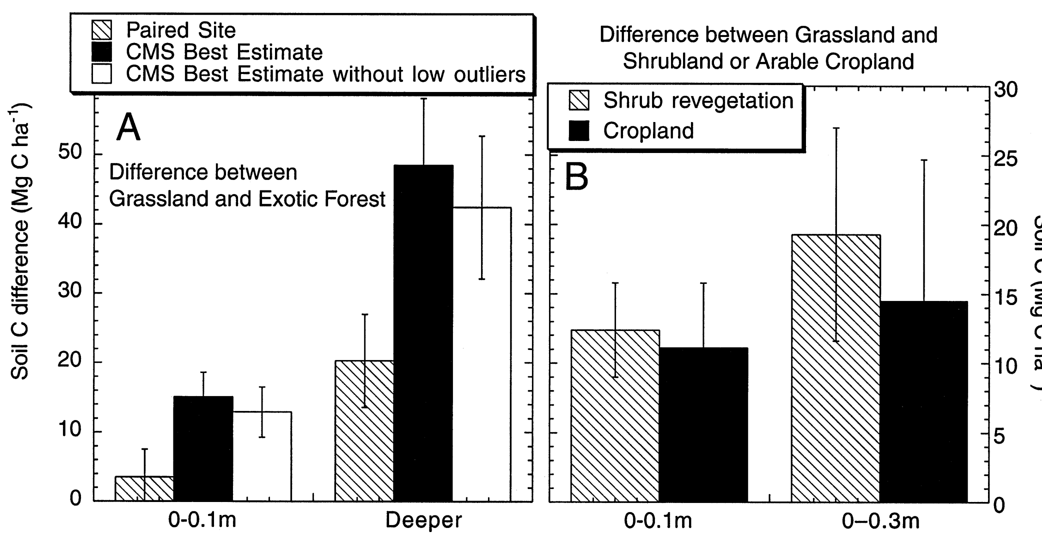

Fig. 3. Soil C differences determined by the soil CMS and paired-site studies. All differences are from grazing land. Error bars indicate 1 standard error. (A) The soil C difference between grazing land and exotic forests from paired-site comparisons and the soil CMS, both with and without low soil C outliers; precipitation × slope factor excluded. (B) Soil C difference between improved grazing land and shrubland or cropland indicated by soil CMS; precipitation × slope factor included.

Table 4. *Some New Zealand vegetation and soil C (0–0.3 m) stocks in 1990, and recent average changes with land-use change*

	Vegetation and soil C stocks				Changes in C stocks (1990–2000)			
Land use	Area (Mha)	Vegetation (Tg)	Soil C (Tg)	Total (Tg)	Areal change (kha yr^{-1})	Vegetation (Tg yr^{-1})	Soil C (Tg yr^{-1})	Total (Tg yr^{-1})
Grazing land	14.0	94	1480 ± 58	1574	−48	−0.3	0	−0.3
Exotic forest	1.3	113	77 ± 23	190	52	4.8 to 6.5[a]	−0.9 ± 0.4[b]	4 to 6
Natural (shrub) vegetation	2.7	283	244 ± 18	431	−8	1.2 to 2.8	0	1.2 to 2.8
Cropland	0.3	1.0	26 ± 3	27	3	0.01	−0.04 ± 0.03	∼0

[a]From Marshall et al. (2001). [b]Calculated assuming 85% forest planted in grazing land.

3.3.4. Testing steady-state assumptions and management impacts. Uncertainties in estimates of soil C changes using the CMS can be attributed to three main sources: (1) the assumption that soil C is at steady state for all land-cover categories; (2) uncertainties in the LUEs; and (3) poor estimates of the areal extent of key land-cover types (and changes in their areas) in the 1990s. For the current analysis, the grazing land category includes both improved (mainly fertilised, legume-based grazing lands) and unimproved grassland (including tussock). Soil C at all depths for these two grassland classes is similar. The LUEs for unimproved grassland compared with improved grassland are –3.1 ± 3.0, 6.6 ± 4.5 and 16.4 ± 13.4 Mg C ha^{-1} for 0–0.1, 0.1–0.3 and 0.3–1.0 m depths, respectively. We used improved grassland as the reference against which to compare soil C changes because (a) almost all contemporary land-cover changes include improved grazing land and (b) a number of published and unpublished reports suggest soil C for this land use can be assumed to be at steady state (Saggar et al., 2001; Section 2.3).

The review of published data from long-term sites, and comparison of soil C in archived and current samples from the same sites, supported the hypothesis that soil C levels in New Zealand's grazing lands are at or near steady-state (Saggar et al., 2001). Furthermore, increases in pasture production following 6- and 15-yr fertiliser additions had no significant effect on soil C concentrations at any of the fertiliser trials (Saggar et al., 2001). These studies were undertaken on established ryegrass (*Lolium perenne* L.) – clover (*Trifolium* sp.) pastures where soil C concentrations were assumed to be at or near steady state. The fact that fertiliser application did not change the soil C concentrations supports the hypothesis that these New Zealand pastures are at or near steady state. Comparison of nationally averaged soil C concentrations in the improved and unimproved grasslands as mentioned above also indicated no significant differences for all depth increments.

The lack of change in soil C as a result of management could have resulted from enhanced decomposition of both soil C and C translocated below ground in the fertilised soils. Fertiliser application to grazing lands alters the quality of root material (narrower C:N and C:P ratios), and increases the quantity of C translocated to the roots in conjunction with higher forage production (Saggar et al., 1997). Recently, Saggar et al. (2000) showed that fertiliser addition not only increased forage production and translocated more C to roots than in non-fertilised grazing lands, but also enhanced soil C decomposition. The net effects of these two processes (change in inputs and soil C decomposition) determine whether there are overall changes in soil C in response to management. Our results indicate that in spite of management-induced changes in C cycling processes, soil C levels in grazing lands remain at or near steady state. Given all these results, we feel justified in making our estimates of LUEs and derived LUFs (Table 2) relative to soil C in grazing lands (nationally estimated at 114 ± 7 MgC ha^{-1} to 0.3 m depth).

3.4. Uncertainty associated with estimating areal extent of key land-cover types

At present, quantifying the areal extent of land-cover change is a major source of uncertainty in using the CMS to quantify soil C changes after 1990, because New Zealand lacks a comprehensive database that provides continuous geo-referenced information over the period 1990–2000.

The VCM (Newsome, 1987) depicts vegetation for the early to mid-1980s, and was used exclusively in the earlier versions of the soil CMS (Scott et al., 2002).

This is a field-checked, compilation survey that is thematically rich, but has low spatial resolution and, having been compiled manually, would be costly to update. It does, however, delineate arable cropping, a significant land cover in the IPCC default methodology, but of minor importance in New Zealand. The advent of the satellite image-derived LCDB (MAF, 2001) in the mid-1990s prompted its adoption as the primary spatial land-cover layer for the CMS. The LCDB delineates 1996/97 land cover with much greater spatial resolution (1:50 000) but at the cost of poorer thematic detail, and does not, for example, delineate arable cropping. The change from VCM to LCDB did not markedly affect LUE and LUF values for the major land-cover types (Table 2). Importantly, however, the LCDB is programmed for updating land cover nationally at intervals roughly coincident with the reporting requirements of the Framework Convention for Climate Change. Neither spatial database, however, depicts land cover precisely at 1990 or 2000.

Using various data sources (section 2.4), we estimated that nearly half (13.61 Mha) the usable land in New Zealand is grazing land used for livestock farming (Table 1). These grazing lands include native tussock grasslands and both improved (perennial ryegrass and white clover) and unimproved grassland (low-fertility grass species). Forests cover about 30% of the land area (Table 1), including about 6.25 Mha in indigenous forests and 1.73 Mha in planted exotic forests (91% of which is *P. radiata*). The area of shrublands is currently estimated as being 2.65 Mha of the land area (Table 1), although diffuse boundaries between grazing land and shrubland, and between forest and shrubland, result in higher uncertainty for the area of this land cover type. Only 1.2% of New Zealand is classified as arable (0.2 Mha) and horticultural (0.1 Mha) land.

Indigenous forests, with 1140 ± 90 Tg vegetation C and shrubland, with 283 Tg vegetation C (Hall et al., 2001), still contain much of New Zealand's terrestrial above-ground C; the C stored in forest litter and soil (to 0.3 m depth) is 1186 Tg. Grazing lands contain the largest store of soil C (1480 ± 60 Tg C to 0.3 m depth) (Table 4).

3.5. Changes in soil C due to land-cover and management changes: 1990−2000

Most contemporary land-cover changes involve the conversion of grazing lands (many of which are highly improved) to other land-cover categories, so the potential for soil C loss is high. Since 1990, about 85% of exotic forests (mainly *P. radiata)* have been planted on improved (45%) and unimproved (40%) grazing lands. These forests rapidly accumulate large amounts of C in the vegetation, ranging from 4.8 to 6.5 Tg C yr^{-1} for all exotic forests (Table 4) during this time (Marshall et al., 2001).

Afforestation of former agricultural land generally causes soil C to decrease, with the largest C losses observed when *P. radiata* is planted on temperate pastures (Paul et al., 2002). Rates of soil C change depend on several factors including previous land use, climate, type of forest established and the time since plantation establishment. The most rapid decrease in soil C occurred within the first 5 yr: soil C declined at 3.64 and 0.63% yr^{-1} for 0–0.1 and 0–0.3 m depth, respectively; following this initial decline, the rate of change decreased, until at age 30 yr soil C may begin to accumulate (Paul et al., 2002). In New Zealand, the accumulation of C after 30 yr may not be observed due to rotation lengths of 20–30 yr. Harvesting may result in short-term soil C losses (Arneth et al., 1998), although long-term changes may be small and difficult to measure (Johnson et al., 2002). Stabilisation of soil C levels is possible as the readily decomposable organic matter from the former grassland inputs declines and more recalcitrant organic matter from the pine increases.

The need to understand historical land-use effects and ecosystem C cycling over several rotations points to a need for process-based models. Attempts to simulate soil C changes associated with the grassland-to-exotic forest transition with CENTURY may predict decreases in soil C (W.T. Baisden and R.L. Parfitt, unpublished results). Despite this, our present understanding of key processes and parameters limits our ability to use models in a predictive sense. Key processes include the decomposition of surface litter, translocation of C and N from forest floor to mineral soil and erosion. Additional work is also needed to better parameterize soil C models for New Zealand's volcanic soils. Future work will focus on partitioning N within the ecosystem because N may limit OM storage and N turnover and availability often limit forest productivity (McMurtrie et al., 2001).

Current national reporting of C accumulation in these forests includes C in the forest floor, but not in the mineral soil. Given the discrepancies between the CMS and the paired-site data, we used the paired-site results to estimate that about 0.9 ± 0.4 Tg C yr^{-1}

could have been lost from mineral soil between 1990 and 2000, as a result of afforestation of grazing land (Table 4). While this loss is small compared with the C gain in the vegetation, it probably offsets any C increase in the forest floor.

Since 1985, following the removal of agricultural subsidies, some areas of former grazing land have reverted to indigenous shrubland. Limited statistics exist for the rates at which this land-cover change is occurring nationally, but it is known to be significant in some regions. Some areas are small and dispersed and dominated by exotic species [e.g. *Ulex europeus* L. (gorse), *Cytisus scoparius* (L.) Link (broom)]. However, the most common indigenous shrubland type is dominated by the indigenous species manuka [*Leptospermum scoparium* J.R. and G. Forst) and kanuka (*Kunzea ericoides* Var. *ericoides* (A. Rich) J. Thompson]. Biomass C accumulates in these shrublands at annual rates varying between 1.7 and 3.8 Mg ha^{-1} yr^{-1} during the active growth phase, depending on site conditions (Trotter et al., 2001). Little change in mineral soil C appears to accompany biomass C accumulation when these shrublands regenerate following fire (Scott et al., 2000), but some C does accumulate in the forest floor. At 23 shrubland sites in South Island, combined fine litter was 8.9 ± 1.1 Mg ha^{-1} and coarse woody debris was 5.2 ± 3.2 Mg ha^{-1} (Coomes et al., 2002).

According to the soil CMS, some soil C loss may occur when shrublands regenerate on abandoned grazing land (Fig. 3), but the few litter C estimates available (Scott et al., 2000; Coomes et al., 2002) suggest litter C may offset mineral soil C loss.

Overall, C accumulation in reverting shrublands is expected to represent a significant and expanding C sink in New Zealand, although obtaining accurate statistics on the rates of change in shrubland area is hampered by differences in thematic and spatial resolution of interdecadal vegetation maps. Although the values in Table 4 suggest that the area of shrublands decreased slightly during the period 1990 to the present, the change is well within current uncertainties for estimates of shrubland area. Reversion of grazing land to shrubland before 1990 means shrublands will have continued to accumulate C after 1990. Our latest national estimate (Table 4) is based on the mean values from biomass sampling over a range of climate and soil fertility conditions (e.g. Scott et al., 2000; Trotter et al., 2001), and is estimated nationally (Table 4) by comparing differences in percentage fractional cover with biomass at these sites (Trotter et al., 2001). These shrublands are the second largest C sink in New Zealand (Table 4). Further reduction in the uncertainties associated with estimating the magnitude of changes in shrubland C accumulation is being addressed in detailed chronosequence studies (e.g. Scott et al., 2000; Trotter et al., 2001) that aim to confirm changes in soil C predicted by the soil CMS.

Cropland has been included in our analysis as it represents a major global land-use change. In New Zealand, however, the contribution of this soil C change to the national C budget is very small (Table 4) due to the negligible area under field crops and horticulture (1.2% of the total land area) and the small rate of conversion. Many studies show soil C changes following conversion between grazing lands and arable land (grain crops) similar to those predicted by the CMS (Fig. 3), but not for horticultural conversion, the dominant land-cover change in this category during the period 1990 to the present (Table 4). No account is taken in this analysis of the potential effects on the terrestrial C budget of soil erosion. These effects are potentially much larger but very uncertain (Tate et al., 2000).

4. Conclusions

Based on the key land-cover changes occurring in New Zealand over the past decade, changes in soil C are not likely to exceed ca. 0.9 Tg C yr^{-1} (excluding the effects of erosion). Most of this change is apparently due to afforestation of grazing land during the 1990s. Although our ability to predict soil C changes at the national scale is strong, our understanding of the key processes controlling soil C changes at specific sites is weak. Further paired-site, and process-modelling, studies are seeking to reduce uncertainties in our estimates of land-cover effects on soil C. More importantly, new methods based on remote sensing are being developed to provide reproducible information on areas of land-cover change through time. This is currently the most important factor contributing to uncertainty in changes to New Zealand's terrestrial C budget during the 1990s.

Our soil CMS goes beyond the IPCC-default methodology in providing a more rigorous statistical basis for quantifying land-cover change effects on soil C and the associated uncertainty. This may be a useful approach for other countries where land-use changes differ from those proposed in the IPCC-default methodology, and where national databases, monitoring networks and historical knowledge are well coordinated.

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