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Soil carbon stocks and land use change: a meta analysis

L. B. GUO* and R. M. GIFFORD†
*CSIRO Plant Industry, GPO Box 1600, Canberra, ACT 2601, Australia, †Cooperative Research Centre for Greenhouse Accounting, GPO Box 475, Canberra, ACT 2601, Australia

Abstract

The effects of land use change on soil carbon stocks are of concern in the context of international policy agendas on greenhouse gas emissions mitigation. This paper reviews the literature for the influence of land use changes on soil C stocks and reports the results of a meta analysis of these data from 74 publications. The meta analysis indicates that soil C stocks decline after land use changes from pasture to plantation (−10%), native forest to plantation (−13%), native forest to crop (−42%), and pasture to crop (−59%). Soil C stocks increase after land use changes from native forest to pasture (+ 8%), crop to pasture (+ 19%), crop to plantation (+ 18%), and crop to secondary forest (+ 53%). Wherever one of the land use changes decreased soil C, the reverse process usually increased soil carbon and *vice versa*. As the quantity of available data is not large and the methodologies used are diverse, the conclusions drawn must be regarded as working hypotheses from which to design future targeted investigations that broaden the database. Within some land use changes there were, however, sufficient examples to explore the role of other factors contributing to the above conclusions. One outcome of the meta analysis, especially worthy of further investigation in the context of carbon sink strategies for greenhouse gas mitigation, is that broadleaf tree plantations placed onto prior native forest or pastures did not affect soil C stocks whereas pine plantations reduced soil C stocks by 12–15%.

Keywords: crop, forest, land use change, meta analysis, pasture, plantation, soil carbon

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Introduction

The terrestrial biosphere can act either as a source or as a sink for atmospheric CO_2, and has been considered to hold the key to the ~2 Gt C year^{-1} discrepancy that persists in estimates of the global carbon cycle designated by the Intergovernmental Panel on Climate Change as 'residual terrestrial uptake' (IPCC 2000). Both the vegetation and the soil may play a part in the residual terrestrial uptake. Therefore, a new challenge in the context of climate change mitigation is the management of terrestrial ecosystem to conserve existing carbon stocks and to remove carbon from the atmosphere by adding to stocks (Malhi *et al*. 1999). Documentation of the results of such management is part of the national greenhouse gas inventory process (IPCC/OECD 1996) that is mandated by the Framework Convention on Climate Change.

Correspondence: L. B. Guo, tel. + 61-2-6246 5267, fax +61-2-6246 5000, e-mail lanbin.guo@pi.csiro.au

Land use change can cause a change in land cover and an associated change in carbon stocks (Bolin & Sukumar 2000). The change from one ecosystem to another could occur naturally or be the result of human activity, such as for food or timber production. Each soil has a carbon-carrying capacity, i.e. an equilibrium carbon content depending on the nature of vegetation, precipitation and temperature (Gupta & Rao 1994). The equilibrium carbon stock is the result of a balance between inflows and outflows to the pool (Fearnside & Barbosa 1998). The equilibrium between carbon inflows and outflows in soil is disturbed by land use change until a new equilibrium is eventually reached in the new ecosystem. During this process, soil may act either as a carbon source or as a carbon sink according to the ratio between inflows and outflows. Some studies have reviewed the effects of certain land use changes on soil carbon stocks, such as forest clearing (Allen 1985), tropical forest clearing (Detwiler 1986), disturbance and recovery (Schlesinger 1986), cultivation (Mann 1986; Davidson & Ackerman

1993), deforestation for pasture (Neil & Davidson 2000), and from cultivation and native vegetation into grasslands (Conant *et al*. 2001). However, the effects of overall land use change need to be reviewed to meet the challenge of managing soil carbon stocks world-wide.

Statistical meta-analytical methods have been developed for quantitative analysis of research results from multiple independent experiments. They have been used successfully with ecological data (e.g. Curtis 1996; Curtis & Wang 1998; Medlyn *et al*. 1999; Conant *et al*. 2001; Johnson & Curtis 2001). These methods usually provide advantages over narrative reviews or quantitative reviews that lack sampling rigor and robust statistical methods (Johnson & Curtis 2001).

The objective of this paper is to review the effects of various land use changes on soil carbon stocks and to summarize the data from the literature using meta analysis.

Methods

Data sources and calculations

We compiled data from the literature on the influence of land use change on soil carbon stocks and characterized each study according to the following categories of land use change: from forest to pasture, from pasture to secondary forest, from pasture to plantation, from forest to plantation, from forest to crop, from crop to plantation, from crop to secondary forest, from pasture to crop and from crop to pasture. The following relevant factors were also included in the compilation: study design, age (time since the conversion), climate data (precipitation and temperature), life zone (tropical, subtropical and temperate), soil texture, soil bulk density, soil sampling depth from soil surface, and other management details (e.g. species, fertilization, irrigation, and rotation).

In the current study, the definition of forest is native forest before it is cleared for other land use. Pasture is land used for grazing purposes including natural grassland. Crop includes lands cultivated for food and fibre products. Secondary forest develops naturally on abandoned land formerly used for other purposes. The main difference between secondary forest and plantation is the human activity that involves establishing the plantation forest.

Literature searches were performed using electronic databases: CAB Abstracts, Biological Abstracts and Web of Science. To be included in our analysis, studies had to report soil carbon concentrations or stocks per unit land area both before and after land use change with means and number of replicates from specific experiment designs (mainly paired-site studies). Reporting of standard deviations was preferred but not essential. In two cases, the authors were contacted for some unpublished data and for missing soil bulk density information to calculate the soil carbon stocks per unit land area (see Acknowledgements). Any study lacking replication was not considered.

In some studies, only soil organic matter (or loss by ignition) or carbon concentration (C_c percentage) was reported. In these cases, soil carbon concentration (C_c%) and total soil carbon stock (C_t t ha^{-1}) are calculated as follows:

$$C_c\% = 0.58 \times OM\%$$

and

$$C_t = BD \times C_c\% \times D$$

where OM% is organic matter (or loss by ignition) as a percentage of soil dry mass, BD is soil bulk density (g cm^{-3}), and D is soil sampling depth (cm). The former equation assumes that soil OM is 58% C (Mann 1986).

For the studies where soil bulk densities were not available, the densities were estimated as follows (Post & Kwon 2000):

$$BD = \frac{100}{\frac{\%OM}{0.244} + \frac{100-\%OM}{1.64}}$$

Despite the problem of lack of independence in some cases, all observations from a paper have been included in this meta analysis because the more conservative approach of using only a single measure sacrifices too much information (Rosenberg *et al*. 2000).

In the current study, soil depth was not adjusted to account for changes in bulk density with land use change unless the authors of the original data had already done so. Not adjusting soil depth may, in some cases, result in a small misestimation of land use effects on soil carbon stocks per unit land area (Post & Kwon 2000).

Meta analysis

A treatment effect size estimator commonly employed in meta analysis is the magnitude of an experimental treatment mean (χ_e) relative to the control treatment mean (χ_c). A typical effect size metric is the response ratio, $r = \chi_e/\chi_c$, or the relative impact on some index (i.e. soil carbon in the current study) following experimental treatment (i.e. land use change in the current study) compared to that in control plots (i.e. original land use in the current study). To be useful statistically, r first must be log transformed such that lr = ln (r) = ln (χ_e) – ln (χ_c). If χ_e and χ_c are normally distributed and χ_c is unlikely to be negative, then lr will be approximately normally distributed with a mean approximately equal to the true response log ratio (Gurevitch & Hedges 2001; Johnson & Curtis 2001).

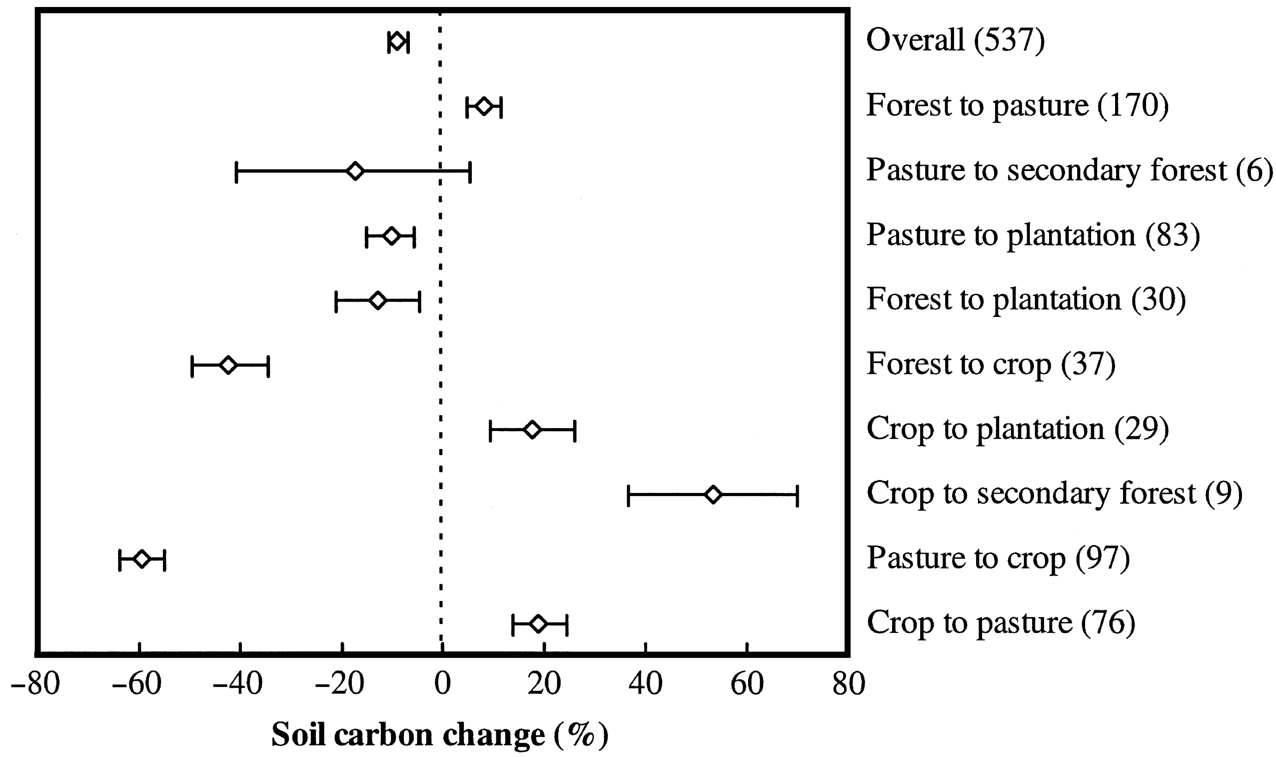

Fig. 1 Soil carbon response to various land use changes (95% confidence intervals are shown and numbers of observations are in parentheses).

Many of the studies did not report any measure of variance for the response variables in which we were interested. Thus, in order to include as many studies as possible, an unweighted meta analysis was used (Johnson & Curtis 2001). Mean effect size for each study was calculated, with bias-corrected 95% confidence intervals (CIs) generated by a bootstrapping procedure (5000 iterations) using Meta-Win software (Rosenberg *et al.* 2000).

An important goal of this meta analysis was to determine whether various land use changes elicited quantitatively different responses in soil carbon stocks. With meta analysis, we tested whether there are significant differences in mean response among various categories of land use change. In a procedure analogous to the partitioning of variance in analysis of variance, the total heterogeneity for a group of comparisons (Q_T) is partitioned into within-class heterogeneity (Q_w) and between-class heterogeneity (Q_b), such that $Q_T = Q_w + Q_b$. The Q statistic follows a chi-square distribution, with k–1 degrees of freedom (Gurevitch & Hedges 2001; Johnson & Curtis 2001).

For analyses of land use change effects, between-group heterogeneity (Q_b) was examined across all data for a given response variable (i.e. C_t) and the overall mean and confidence interval (CI) were presented. Then each group was further analysed for the effects of other relevant factors. Means were considered to be significantly different from one another if their 95% CIs were nonoverlapping, and were significantly different from zero if the 95% CI did not overlap zero (Gurevitch & Hedges 2001).

Results

Five hundred and thirty-seven observations from 74 publications were included in the database for the current study (Appendix 1). The database covers 16 countries, but most studies were in the following four countries: Australia, Brazil, New Zealand and USA. Paired-site design was the main experimental design. There were also some studies with chronosequence and repeated-sampling designs.

As an overall average across all land use change categories and influential variables examined, land use change reduced soil C stocks by 9% (Fig. 1). However, not all categories of land use change involved decreases in soil C stocks. Soil C stocks significantly increased after the conversion from forest to pasture (+ 8%), crop to plantation (+ 18%), crop to secondary forest (+ 53%), and crop to pasture (+ 19%). Soil C declined after the conversion from pasture to plantation (−10%), forest to plantation (−13%), and particularly from forest and pasture to crop (−42% and −59%, respectively). The highest

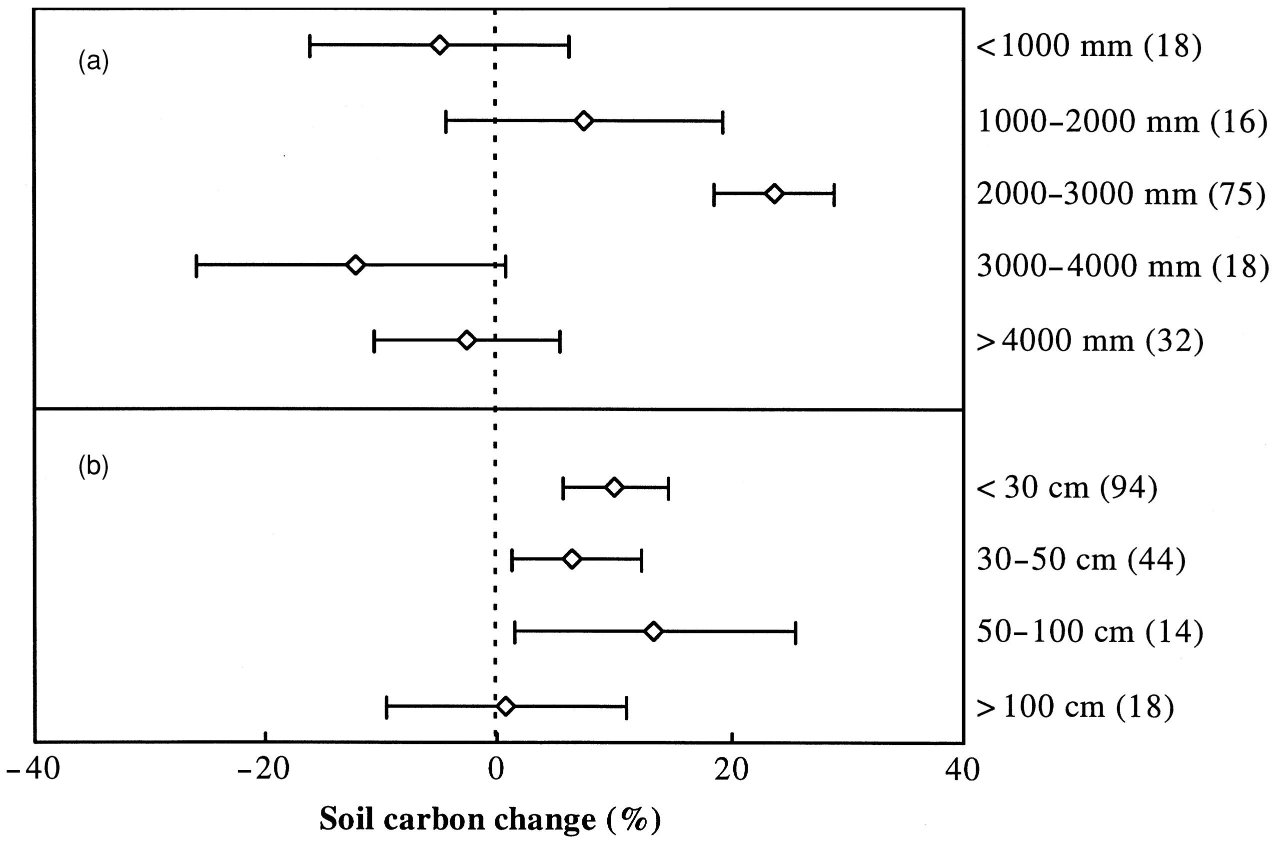

Fig. 2 The effects of precipitation (a) and soil sampling depth (b) on soil carbon after land use change from forest to pasture (95% confidence intervals are shown and numbers of observations are in parentheses).

soil C loss occurred following land use change from pasture to crop. Wherever one of the land use changes decreased soil C (e.g. pasture to crop), the reverse process (e.g. crop to pasture) increased soil carbon, and *vice versa* except after the change from pasture to secondary forest. The latter was not statistically significant possibly owing to the low number of observations for that transition.

The data in Fig. 1 were further analysed in relation to the relevant factors within those land use categories having a large number of observations (viz. forest to pasture, pasture to plantation, forest to plantation, forest to crop, pasture to crop and crop to pasture). The factors found to exhibit significant influence on soil carbon stocks are as follows.

Forest to pasture

Within the observations for the land use change from forest to pasture, precipitation and soil sampling depth each had significant effects on the change in soil C stocks (Fig. 2). Clearing of forest for pasture in areas with 2000–3000 mm precipitation sequestered significantly more soil C stocks (+ 24%), but not in other areas with either lower or higher rainfall.

Soil C stocks increased by 7–13% if sampling depth was less than 100 cm, but there was no change below 100 cm. However, the data did not provide any evidence that within the top meter, the relative soil C change was more marked in the top soil than further down the profile.

Pasture to plantation

Tree type and precipitation affected the magnitude of soil C stocks after forest was planted onto pasture (Fig. 3). Planting broadleaf trees into pasture had little effect on soil carbon stock. There was no significant difference between N-fixing trees (i.e. legume species and *Casuarina*) and other broadleaf species (e.g. *Eucalyptus* and *Populus*). In contrast, planting conifer trees (mainly *Pinus radiata*) significantly reduced soil C stocks by 12%.

The conversion had little effect on soil C stocks in the lower rainfall (<1200 mm) areas, but significantly reduced soil C stocks in higher rainfall areas, especially in the areas with precipitation >1500 mm (−23%).

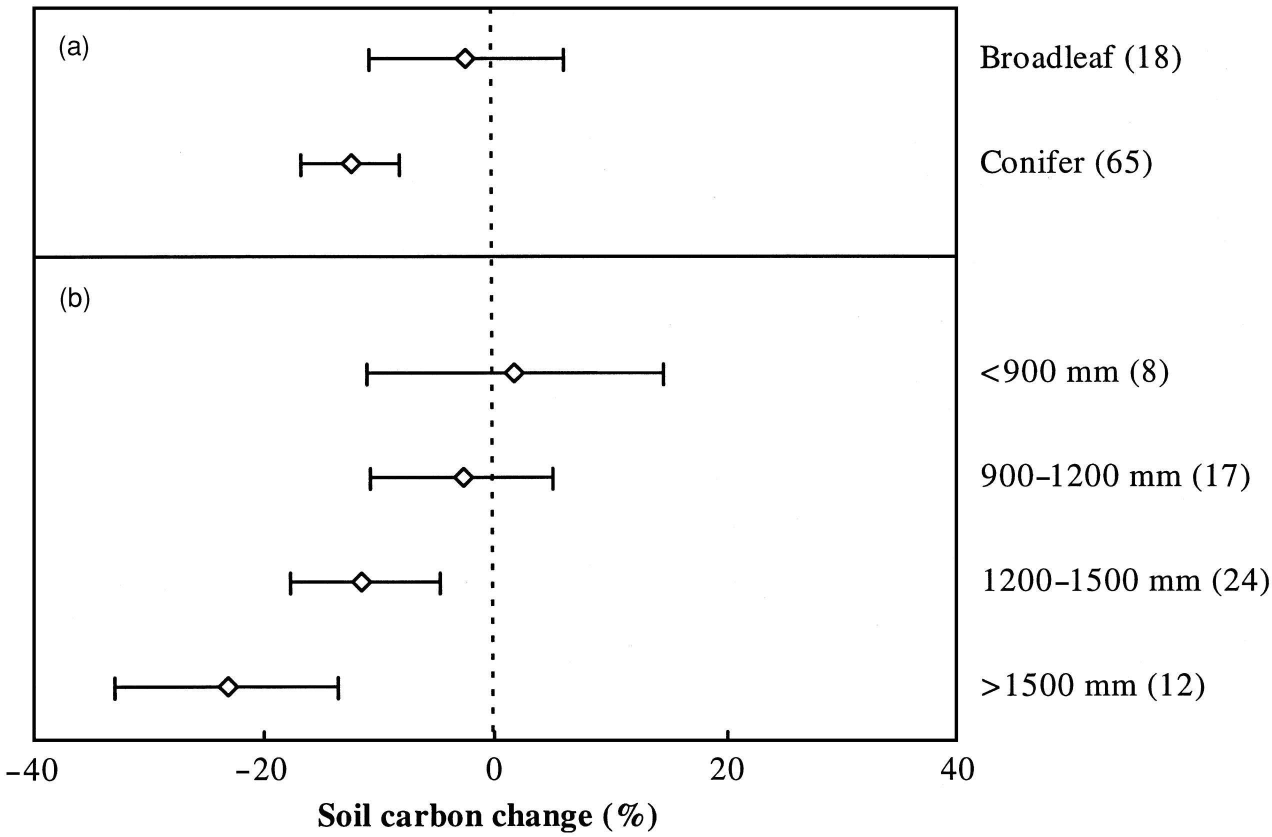

Fig. 3 The effects of tree type (a) and precipitation (b) on soil carbon after land use change from pasture to plantation (95% confidence intervals are shown and numbers of observations are in parentheses).

Forest to plantation

As with the conversion from pasture to plantation, tree type and precipitation had effects on soil C stocks after plantations were established on prior native forest. The trends were identical (Fig. 4). Planting broadleaf trees had little effect on soil carbon, but planting conifer trees significantly reduced soil C stocks by 15%. Soil C was released (−56%) only in the areas with precipitation > 1500 mm after land use changed from forest to plantation. Besides species and precipitation, plantation age also had significant effects on the soil C stocks following the land use change. Carbon stocks were reduced by about 20% when the plantations were less than 40 years old. However, soil C was restored to the original level in plantations more than 40 years old.

Forest to crop

Of the factors analysed, only soil sampling depth influenced the soil C results after land use change from forest to crop (Fig. 5). Soil C stocks decreased by about 50% if sampling depth was less than 60 cm, but did not change below 60 cm. Within the top 60 cm, the data did not show evidence of a stronger response in the top soil.

Pasture to crop

Of the factors examined, only precipitation and age had effects on soil carbon stocks following conversion of pasture to cropland (Fig. 6). The change always reduced soil C stocks by about 50% or more, but the highest carbon loss (−78%) was found in the areas with 400–500 mm precipitation. Also, the magnitude of the loss of soil C stocks peaked 30–50 years after conversion at 85% loss, recovering somewhat thereafter.

Crop to pasture

Of the factors analysed, only soil sampling depth influenced the magnitude of increase of soil C stocks after cropland was planted to pasture (Fig. 7). In this case the deeper the sampling depth, the less was the fractional effect of the pasture on soil C stocks.

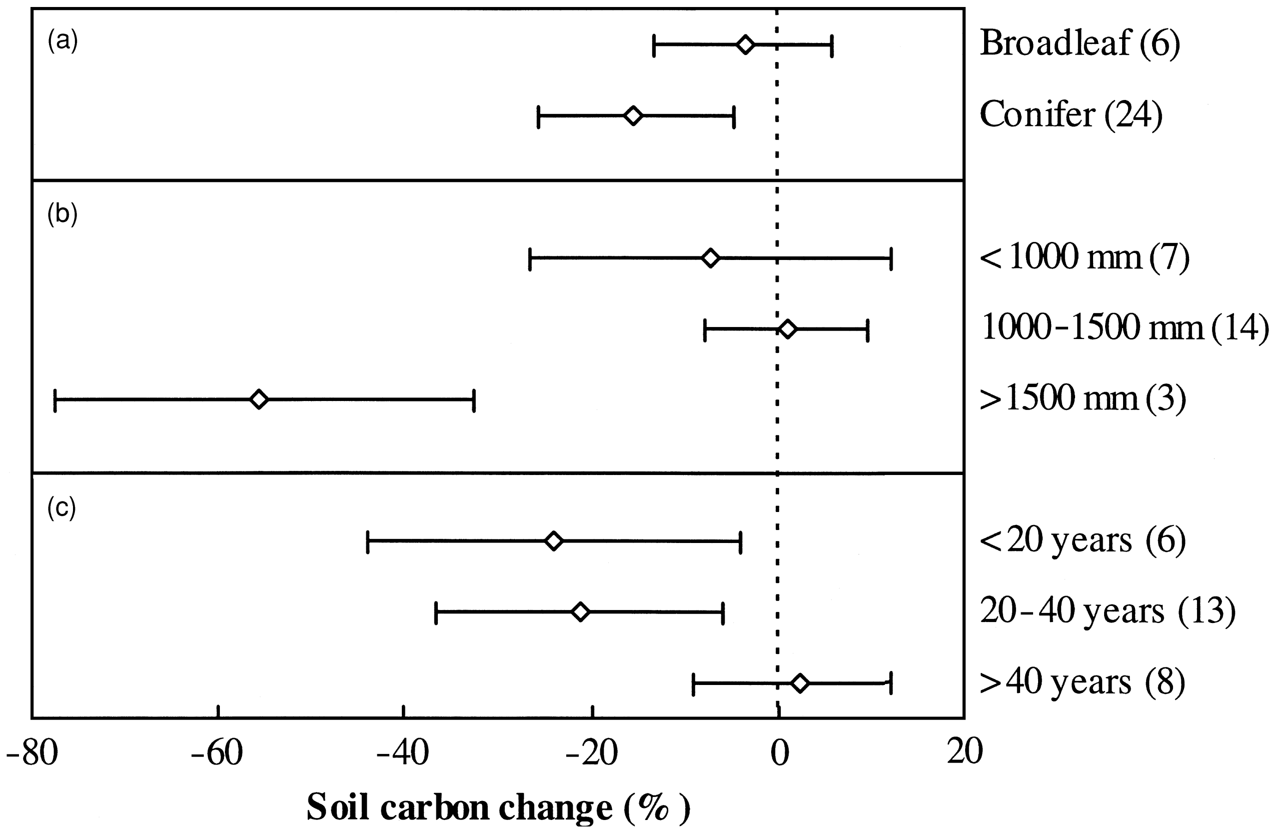

Fig. 4 The effects of tree type (a), precipitation (b) and age (c) on soil carbon after land use change from forest to plantation (95% confidence intervals are shown and numbers of observations are in parentheses).

Discussion

Forest to pasture

When forest is cleared to establish pasture, considerable aboveground C in vegetation is lost, but it is not necessary that there be declines in soil organic C (Post & Kwon 2000). Pasture established following clearing of forest has greater potential soil organic C stock than it has following crop and, in the long term, grass management systems have nearly equivalent potential to store soil organic C as forest (Franzluebbers *et al*. 2000). Stevenson (1982) indicated that the organic matter content of grassland soils was substantially higher than for forest soils if other factors were constant. Hence, soil C stocks could be higher under natural grasslands than under natural forests. For example, Tate *et al*. (2000) reported that total soil profile C stock was 13% higher in the grassland than in the forest they studied (19.9 vs. 16.7 kg m^{-2}). The differences mainly resulted from more recalcitrant soil C stocks in the grassland (5.3 vs. 3.0 kg m^{-2}) than in a montane beech (*Nothofagus sloandri* var. *cliffortiodes*) forest and adjacent tussock grassland (*Chionochloa pallens*) in New Zealand. They suggested that the different patterns of soil C accumulation in these ecosystems have resulted from differences in plant carbon inputs, soil aluminium, and soil physical characteristics, rather than from differences in soil mineral weathering or texture.

Unlike annual crops, pasture grasses continuously maintain a cover of vegetation on the soil, reduce soil temperatures, and sometimes have high productivity and turnover rates that add organic matter, particularly from belowground, to the soil (Brown & Lugo 1990). Because pastures are generally not cultivated, the loss of carbon from pasture soils is usually less than 25% of the initial carbon contained in the top 100 cm under forest, but the findings are extremely variable (Houghton 1995), e.g. increased (e.g. Moraes *et al*. 1996; Neill *et al*. 1997) and decreased (e.g. Detwiler 1986; Fearnside & Barbosa 1998). Meta analysis can explore such variability. The results from the analysis showed that soil C stocks would increase on average by 8%, rather than lost, after land use changed from forest to pasture (Fig. 1). Hence, soil carbon, on average, may increase following land use change from forest to pasture although variation exists.

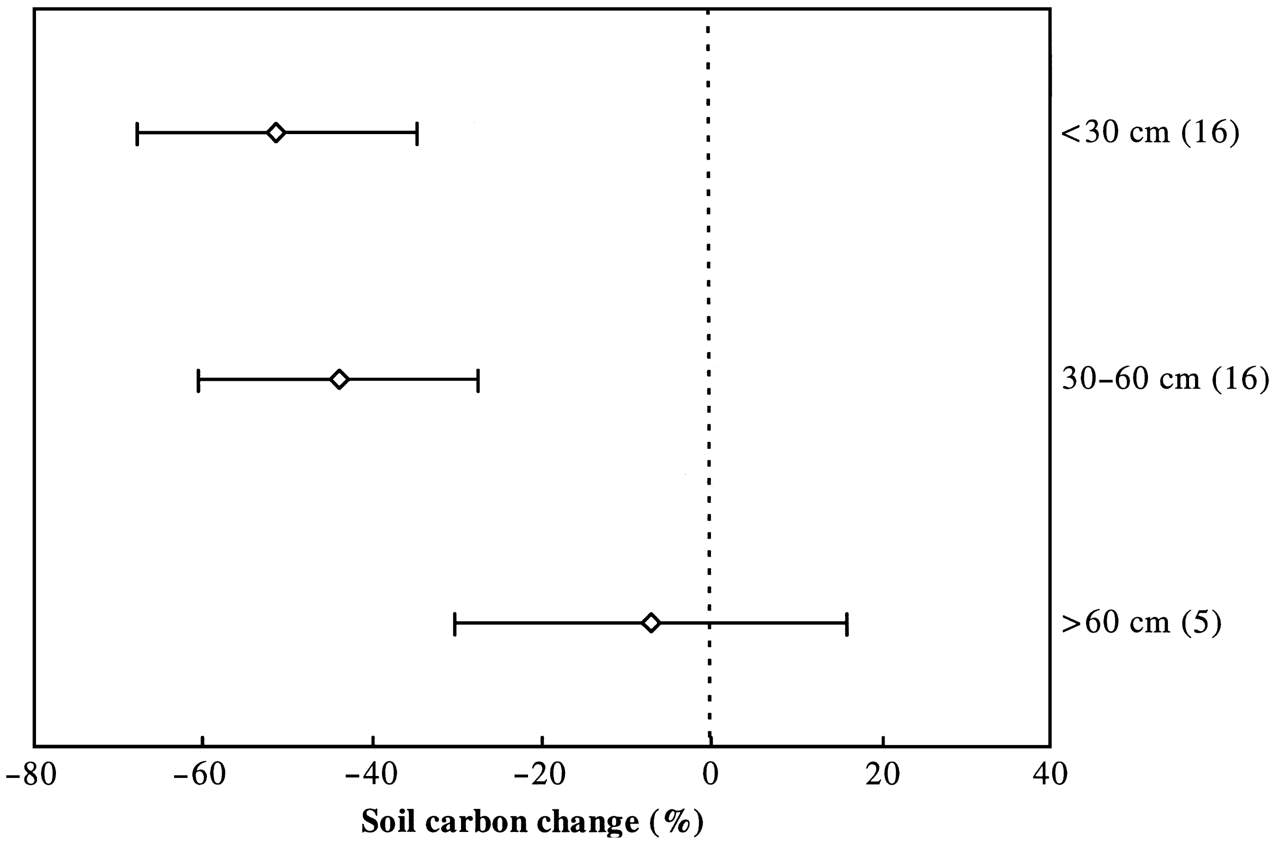

Fig. 5 The effects of sampling depth on soil carbon after land use change from forest to crop (95% confidence intervals are shown and numbers of observations are in parentheses).

Yakimenko (1998) reported a rapid increase in soil organic matter in the topsoil during grassland formation just after deforestation. The higher level of soil organic C stock observed in the soils of all studied grasslands in comparison with the woodlands, may be related not only to the more intensive humus formation in the soils with a great amount of fine grass roots. He proposed that reduced water and gaseous exchange in meadow soils due to the dense root net may also reduce the rates of decomposition in the meadow ecosystems.

Conant *et al*. (2001) concluded that conversion of native land cover (mainly rain forests) to grassland led to increased soil C stocks for nearly 70% of the studies. Lugo & Brown (1993) indicated that there was often as much or more soil organic C in many tropical pastures as in adjacent forests across a range of life zones as pastures could accumulate high amounts of soil organic C relative to adjacent forest. This was particularly the case in pastures younger than 20 years. However, the meta analysis of the current study did not show an age effect on soil C stocks after clearing forest for pasture.

Clearing forest for pasture increased the soil C stocks by 24% in the areas with 2000–3000 mm (Fig. 2a), but had no effects in areas with rainfall less than 2000 mm or more than 3000 mm. Perhaps conversion of forest to pasture in areas with >3000 mm annual rainfall led to initial topsoil erosion and associated loss of C which was only gradually recovered under pasture. Hence precipitation should be considered in managing soil C stocks during the land use change from forest to pasture.

Soil sampling depth had effects on the results. Clearing forest for pasture increased the soil C stocks only in the top 100 cm (Fig. 2b). This result may be related to the difference of root distribution between the two land uses. In pasture, less carbon may be contributed by roots into deeper layers.

Other factors may also affect soil carbon stock after land use change from forest to pasture. For example, Fearnside & Barbosa (1998) found in their review that soil C decreased if pastures were poorly managed.

Pasture to secondary and plantation forests, and forest to plantation

The reverse of land use change from forest to pasture can be divided into two groups: from pasture to secondary

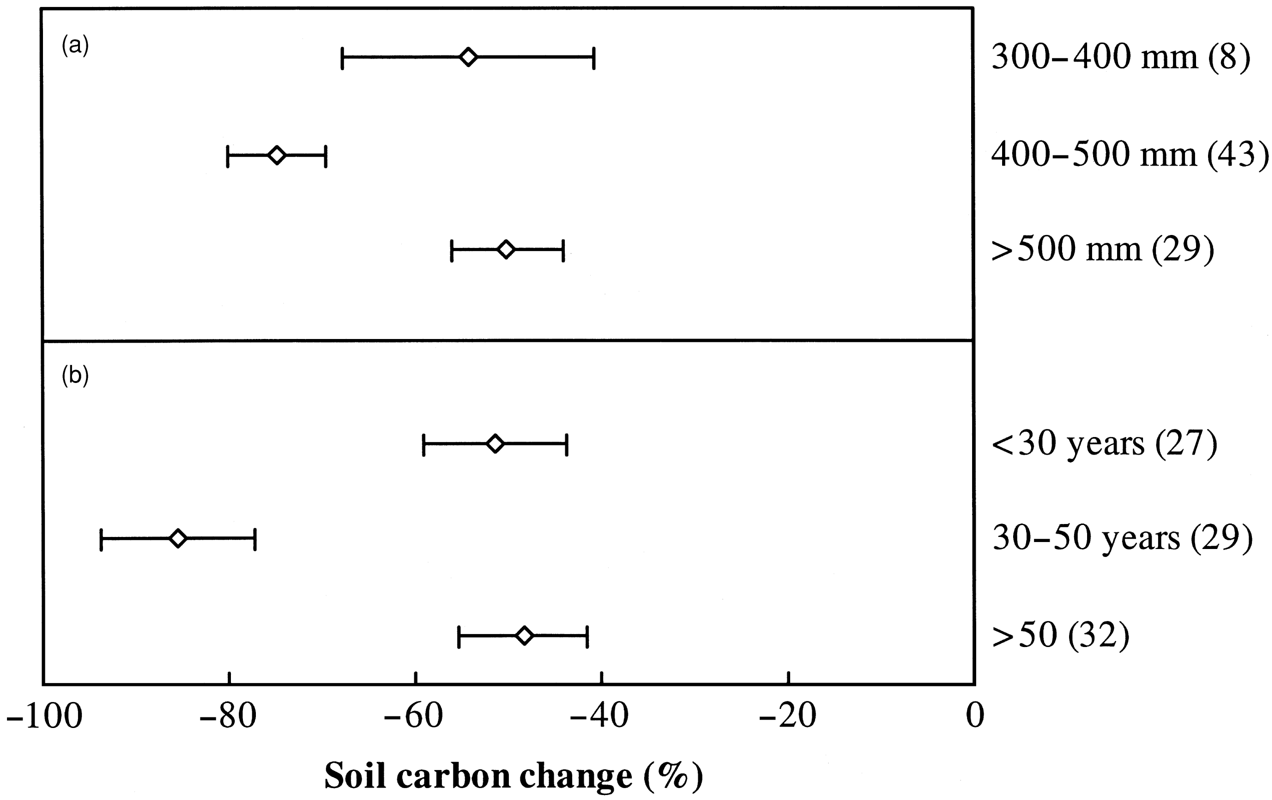

Fig. 6 The effects of precipitation (a) and age (b) on soil carbon after land use change from pasture to crop (95% confidence intervals are shown and numbers of observations are in parentheses).

forest and to plantation forest. In the former case, forest succeeds naturally without continuing human activity involved, while in the latter, human activities, such as site preparation, tree planting, etc., are involved. Both processes reduced the soil C stocks, but not significantly so for natural succession.

In their review on the vertical distribution of soil organic carbon world-wide, Jobbagy & Jackson (2000) found that the relative distribution of soil organic carbon in the top meter of soil was deeper in grasslands (42% in the uppermost 20 cm) than in forests (50% in the uppermost 20 cm). Therefore, aboveground inputs and relatively low decomposability in forests could increase soil organic matter in surface soils compared to grasslands. Hence, tree roots are less important sources of organic matter than grass roots because much of the tree root system lives for many years. The annual turnover of organic matter from dying tree roots is therefore smaller than from grass roots. Post & Kwon (2000) concluded that growth of woody plants resulted in a decrease in soil organic carbon, despite the fact that woody plants produced a greater amount of more recalcitrant material. Woody plants may be less effective than perennial grasses in some environments at storing carbon in soil. Woody plants deposit a larger fraction of total inputs than grasses on the surface where decomposition might involve less formation of soil organic matter. The current meta analysis supports their conclusion. Hence, trees planted onto pasture land reduced soil C stocks by 10% rather than increasing it. Similarly, the conversion from forest to plantation also reduced soil C stocks (Fig. 1). Soil disturbances involved in establishment of plantations may result in decomposition of soil organic matter and losses of carbon occurring at different rates in different parts of the soil profile. It may be also related to the change from a multi-storey forest to a single storey plantation lacking understorey.

Jenny (1980) indicated that of several variables influencing the capacity of soils to store soil organic matter, mean annual precipitation exerts the greatest influence. Higher rainfall, despite the advantage of moister soil for decomposition, is thought to be associated with a larger soil organic C pool and greater leaching of C to the deep profiles (Jenny 1980; Post *et al.* 1982). On the other hand, site preparation and tree planting can disturb soil structure and may break soil aggregates. Then, some part of soil C may lose its physical protection after land use changes from forest and pasture to plantation. Therefore,

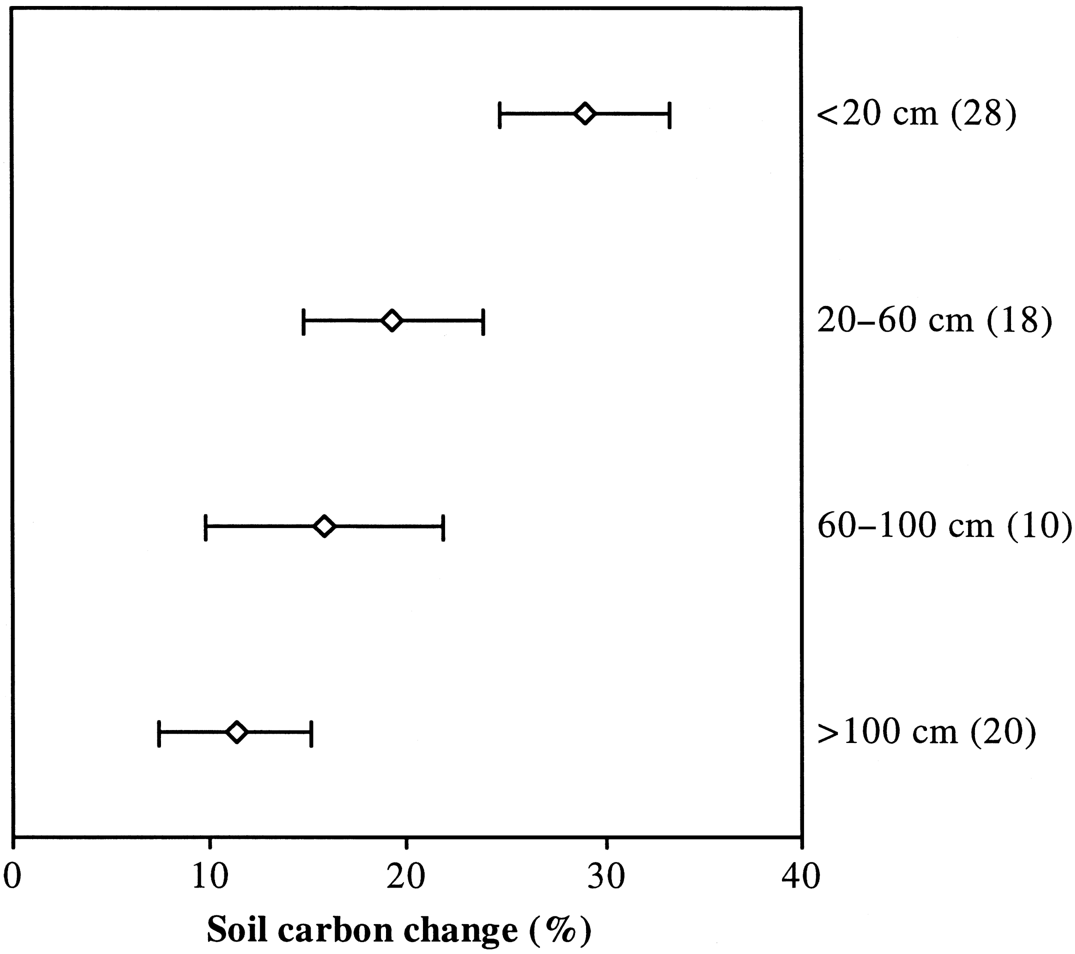

Fig. 7 The effects of sampling depth on soil carbon after land use change from crop to pasture (95% confidence intervals and number of observations are shown in parentheses).

C leaching may play some role in the reduction of soil C stocks in the areas with high annual precipitation (Figs 3 and 4). However, this was not evident from analysing data on soil sampling depth.

Soil organic carbon accumulation in tree plantations is affected by species as some species produce and accumulate more litter or roots than others and these differential rates of organic matter production eventually influences soil organic carbon (Lugo & Brown 1993). After the conversion both from pasture and forest to plantation, species affected soil C stock changes (Figs 3 and 4). Planting conifer trees (mainly *Pinus radiata*) onto pasture or forest land significantly reduced soil C stocks. Other studies (e.g. Turner & Lambert 2000), not included in the database, also showed that soil C stocks decrease under conifer plantation. However, planting broadleaf trees had little effect on soil carbon. Carbon storage in forest soils is affected by forest type (species, deciduous, evergreen) and site quality (Lal *et al*. 1995). Similarly, site quality and forest type also affect soil C stocks under plantation forest. Each species has a different carbon allocation strategy that results in a different pattern, rate, quality and quantity of organic carbon input to the soil (Lugo & Brown 1993). The difference between conifer and broadleaf trees may be related to their individually inherited strategy to allocate assimilated carbon below ground. Root biomass and turnover may be more important determinants for the accumulation of soil organic carbon in forests than aboveground litter input, at least in the short term (Lugo & Brown 1993). For example, Coleman *et al*. (2000) found that pine fine-root production was only 2.9% of that of poplar during their three year study.

Among broadleaf trees, N-fixing species may sequester more soil C stocks than the other species owing to the extra nitrogen input. For example, Resh *et al*. (in review) found that N-fixing plantations sequestered more soil C than eucalypt plantations. However, Ewers *et al*. (1996)

reported that soils from *Eucalyptus saligna* stands had more total soil carbon, but less total nitrogen than soils from *Albizia falcataria* (a N_2-fixing species) stands. The meta analysis in the current study revealed no significant difference in soil C stocks between N-fixing species (legume trees and *Casuarina*) and other species (mainly *Eucalyptus* and *Populus*). On the other hand, fertilization during or after the land use change from pasture or forest to plantation did not have any significant effects on soil C stocks either (data not shown). Similarly, fertilization during or after the land use change from other ecosystems to pasture also had no significant effects on soil C stocks (data not shown). Hence, fertilization or planting N-fixing species can increase biomass production in plantation forest and potentially increasing carbon input to the soil, but it may also enhance decomposition. Moreover, fertilizer application may reduce the relative allocation of carbon belowground (Haynes & Gower 1995). These may be the reasons why fertiliser does not necessarily increase soil carbon stocks.

The time since conversion influenced soil C stocks in the change from forest to plantation (Fig. 4c). Soil C stocks under plantation forest could be restored to the original level under native forest, but it requires at least 40 years. However, no such restoration would occur after land use change from pasture to plantation where age had no effect on soil carbon.

There was a decrease of organic carbon in the upper mineral soil when land in pasture was converted to *Pinus radiata*; however, this decrease was largely compensated by accumulations of organic carbon in the litter layer under the pine (Parfitt *et al*. 1997). Hence, plantations would not reduce soil C stocks if carbon in the litter layer on the top of the soil surface were included.

Forest and pasture to crop

Soil C was significantly lost after conversion from forest or pasture to crop (Fig. 1). This result is comparable with some narrative and quantitative reviews on such land use changes (e.g. Allen 1985; Mann 1986; Schlesinger 1986; Davidson & Ackerman 1993).

Houghton (1995) indicated that clearing forests for new agricultural land causes a release of carbon to the atmosphere. The carbon initially held in trees and other vegetation is released through burning or through decomposition of above- and belowground plant material left in the soil at the time of clearing. Even if the productivity of the new agricultural land is as high as it was in the forest, less of the crop production accumulates as litter; most of it is harvested and subsequently consumed or respired. However, in estimating the CO_2 released from conversion of forest to agricultural land a distinction needs to be clearly made between pastoral agriculture, arable agriculture and mixed farming involving crop/pasture sequences.

The level towards which organic pools tend under cultivation suggested that the decay rates of soil carbon were one order of magnitude higher under cultivation than under forest. Soil organic matter can thus be considered as de-protected under cultivation (Balesdent *et al*. 1998). Hence, the process of the conversion from forest or pasture to crop and management afterward reduces carbon input from litter and enhances the carbon output via breaking the protection of soil organic matter. Reicosky (1997) indicated that there was potential for using the soil as a sink for carbon through improved soil and tillage management even though intensive tillage could cause large gaseous losses of carbon. For example, Aslam *et al*. (1999) found that adoption of no-tillage could protect soils from biological degradation and maintain soil quality as compared with plough tillage management after land use change from pasture to crop. However, there are not enough data available to perform a meta analysis of the land use change from pasture or forest to no-tillage crop.

Loss of soil carbon after the conversion from forest to crop varies significantly with soil sampling depth (Fig. 5). The conversion had no influence on soil C stocks beyond 60 cm depth, but it significantly reduced carbon above the 60 cm depth.

After the conversion from pasture to crop, more soil C was lost from land with 400–500 mm precipitation (− 75%) than from lands with 300–400 mm (− 54%) and > 500 mm (− 50%) (Fig. 6a). The reason is not clear. On the other hand, more carbon loss was found 30–50 years after the conversion than < 30 years and > 50 years. 'Stable' carbon may play some role in this age effect. The relatively stable part of soil carbon from previous land use (pasture) should be released gradually, but the part from current land use (crop) may be gradually accumulated at a slower rate. Therefore, the equilibrium in the new land use could only be reached after 50 years. This hypothesis could be tested via the conversion from C3/C4 plant ecosystems.

Crop to plantation, secondary forest and pasture

The reverse process of the conversion from forest or pasture to crop, i.e. from crop to plantation, secondary forest or pasture, significantly sequestered carbon from the atmosphere (Fig. 1). However, this process could not fully recover the amount of carbon lost from soil during the process in other direction except for the secondary forest. Age in secondary forest may play the key role in that. For example, Brown & Lugo (1990) found that there was a general pattern of increasing soil C stocks with increasing age of secondary forests, the oldest secondary forest (≈50 years old) having approximately the same soil carbon content as the mature forest in subtropical life zones. On the other hand, the lack of soil disturbance during the land use change, and the multistorey structure above ground and possibly also underground could play some role in it too.

Carbon sequestration after the conversion from crop to pasture varied with soil sampling depth (Fig. 7). The deeper the depth, the less the carbon sequestration. This shows that the top soil is more active at sequestering carbon from atmosphere after the land use change. However, pasture also caused substantial C accumulation below 100 cm depth.

Post & Kwon (2000) indicated that there are many factors and processes that determine the direction and rate of change in soil organic content when vegetation and soil management practices are changed. Ones that may be important for increasing soil organic carbon storage include: (i) increasing the input rates of organic matter; (ii) changing the decomposability of organic matter inputs that increase the light fraction organic carbon (in particular); (iii) placing organic matter deeper in the soil either directly by increasing belowground inputs or indirectly by enhancing surface mixing by soil organisms; and (iv) enhancing physical protection through either intra-aggregate or organomineral complexes. Conditions favouring these processes occur generally when soils are converted from cultivated use to permanent perennial vegetation, such as from crop to pasture, plantation and secondary forest in the current study. In contrast, conditions not favouring these processes should occur when soils are converted from permanent perennial vegetation to cultivated use, i.e. from forest and pasture to crop in the current study.

Organic carbon (C_{org}), microbial carbon (C_{mic}), and the C_{mic}:C_{org} ratio were consistently higher in pasture soils, suggesting that the belowground carbon input under pastures was greater than in the equivalent soils under native forests, exotic forest or arable cropping (Sparling *et al*. 1992). Hence, high root production by grasses may explain why pastures accumulate so much soil organic carbon (Cerri *et al*. 1991). On the other hand, episodic grazing or cutting of pastures may have a soil carbon 'pumping action' owing to the rapid death of roots following each defoliation event followed by root regrowth as the pasture sward re-establishes. Under certain conditions, grazing can lead to increased annual net primary production over ungrazed areas (Conant *et al*. 2001). Most pasture plants (≈80%) are perennial and have well developed root systems that are used as a carbon storage of new growth in spring or after grazing (mowing). Hence, the relative belowground translocation of assimilated carbon by pasture plants can reach up to 80% (including C autotrophically respired by roots) but up to only 60% by trees (Kuzyakov & Domanski 2000). At the same time, grasslands have high inherent soil organic matter content that supplies plant nutrients, increases cation exchange and water holding capacities (Miller & Donahue 1990). Therefore, soil C stocks will be increased if the land is converted from other land uses to pasture, and reduced if from pasture to other land uses, especially to crop.

Since the world database available for this meta analysis is quite small, the criteria for data inclusion had to be rather relaxed. Hence the results may be biased in some circumstances by dominance of data from certain regions or authors. Accordingly, it is best to regard all the results deemed 'significant', as hypotheses requiring targeted investigation using a common methodology. Some of these hypotheses are more solid than others. One of the unexpected findings was that pine plantations tended to reduce soil C stocks after land use change whereas broadleaf species did not. From that observation we propose a secondary explanatory hypotheses that pine roots may have a property of stimulating the decomposition of litter and soil organic matter (e.g. breaking soil structure for exploiting nutrients in soil organic matter), and reducing carbon input into soil (e.g. removing a potential 'soil carbon pumping action' of grazed pastures, and placing a greater fraction of litter onto the soil surface rather than below ground).

Conclusions

The main hypotheses suggested by this meta analysis of the world literature on changes in soil C stocks following land use changes are:

1. After native forests are cleared for pastures, soil C stocks do not generally decline and tend to increase in areas receiving 2000–3000 mm year^{-1} of annual precipitation.
2. When established pastures switch to forest, soil C stocks decline under pine plantations but are unaffected by either broadleaf tree plantations or naturally regenerated secondary forest.
3. When native forest is cleared for plantation forestry, soil C stocks are unaffected by broadleaf plantations and by pine plantations in low rainfall areas but decline where rainfall exceeds 1500 mm year^{-1}.
4. When native forest is cleared to cropland, soil C stocks are halved in the topsoil but not affected at depth.
5. When cropland reverts to forest, there is recovery in soil C stocks. This is a partial recovery for plantation forestry and can ultimately be a full recovery for naturally regenerated secondary forest.
6. When cropland is placed under pasture, soil C stocks increase to below 100 cm depth but the fractional increase decreases with depth.

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Appendix 1

References included in the database for meta analysis (74 studies)

Land use change	Location	Study design	Author(s)
Forest to plantation	Nigeria	Paired-site	Aborisade & Aweto (1990)
Pasture to plantation	New Zealand	Paired-site	Alfredsson *et al.* (1998)
Forest to crop	Nigeria	Paired-site	Aweto (1988)
Forest to plantation	Nigeria	Paired-site	Aweto & Ishola (1994)
Forest to crop	Nigeria	Paired-site	Aweto & Obe (1993)
Forest to Crop	France	Paired-site	Balesdent *et al.* (1998)
Crop to plantation	USA	Paired-site	Bashkin & Binkley (1998)
Crop to pasture	Australia	Paired-site	Bell *et al.* (1995)
Crop to plantation	USA	Repeated-sampling	Binkley & Resh (1999)
Pasture to crop	USA	Paired-site	Blank & Fosberg (1989)
Pasture to crop	USA	Chronosequence	Bowman *et al.* (1990)
Forest to pasture	Puerto Rico	Paired-site	Brown & Lugo (1990)
Forest to pasture	Australia	Paired-site	Bruce (1965)
Crop to pasture, pasture to crop	USA	Paired-site	Burke *et al.* (1995)
Pasture to crop	USA	Paired-site	Cambardella & Elliott (1992)
Forest to pasture	Brazil	Chronosequence	Cerri *et al.* (1991)
Pasture to crop	Australia	Paired-site	Chan *et al.* (1995)
Pasture to plantation	New Zealand	Paired-site	Chen *et al.* (2000)
Pasture to Plantation	New Zealand	Paired-site	Condron & Newman (1998)
Forest to pasture	Costa Rica	Chronosequence	Dam *et al.* (1997)
Pasture to plantation	New Zealand	Paired-site	Davis (1994)
Pasture to plantation	New Zealand	Paired-site	Davis (2001)
Pasture to plantation	New Zealand	Paired-site	Davis & Lang (1991)
Pasture to crop	Ethiopia	Paired-site	Duffera & Robarge (1996)
Forest to pasture	Brazil	Paired-site	Eden *et al.* (1990)
Forest to pasture	Brazil	Paired-site	Eden *et al.* (1991)
Forest to pasture	Brazil	Chronosequence	Feigl *et al.* (1995)
Forest to pasture	Mexico	Chronosequence	Garcia-Oliva *et al.* (1994)
Crop to pasture, pasture to crop	USA	Paired-site	Gebhart *et al.* (1994)
Pasture to plantation	New Zealand	Paired-site	Giddens *et al.* (1997)
Forest to pasture, pasture to plantation	Australia	Paired-site	Gifford (2000)
Forest to pasture, pasture to plantation	Australia	Paired-site	Gifford & Barrett (1999)
Crop to plantation	USA	Repeated-sampling	Gilmore & Boggess (1963)
Forest to pasture	Kyrgyzia	Paired-site	Glaser *et al.* (2000)

Land use change	Location	Study design	Author(s)
Forest to plantation	New Zealand	Paired-site, Chronosequence	Goh & Heng (1987)
Forest to crop, forest to pasture	Australia	Paired-site	Graham *et al.* (1981)
Forest to crop	Canada	Paired-site	Gregorich *et al.* (1995)
Crop to plantation, pasture to plantation	USA	Paired-site	Grigal & Berguson (1998)
Pasture to plantation	New Zealand	Repeated-sampling, Paired-site	Guo (1998)
Forest to plantation	Australia	Paired-site	Hamilton (1965)
Crop to plantation	USA	Paired-site	Hansen (1993)
Forest to crop	Japan	Paired-site	Higuchi & Kashiwagi (1993)
Crop to secondary forest, forest to crop	USA	Paired-site	Huntington (1995)
Crop to pasture, pasture to crop	USA	Paired-site	Ihori *et al.* (1995)
Pasture to plantation	Germany	Repeated-sampling	Jug *et al.* (1999)
Forest to pasture, forest to crop	Costa Rica	Paired-site	Krebs (1975)
Forest to crop	Brazil	Paired-site	Lepsch *et al.* (1994)
Crop to secondary forest, forest to crop, pasture to secondary forest	Puerto Rico, USA	Chronosequence	Lugo *et al.* (1986)
Forest to plantation	Brazil	Paired-site	McGrath *et al.* (2001)
Forest to pasture	Brazil	Chronosequence	Moraes *et al.* (1996)
Forest to pasture	Brazil	Chronosequence, Paired-site	Neill *et al.* (1997)
Forest to pasture	Brazil	Chronosequence	Neill *et al.* (1998)
Pasture to plantation	Australia	Paired-site	Noble *et al.* (1999)
Pasture to plantation	New Zealand	Paired-site	Parfitt *et al.* (1997)
Pasture to plantation	New Zealand	Paired-site	Perrott *et al.* (1999)
Crop to plantation, pasture to plantation	USA	Paired-site	Resh *et al.* (in review)
Forest to crop, forest to pasture	Ecuador	Paired-site	Rhoades *et al.* (2000)
Pasture to crop	Italy	Paired-site	Riffaldi *et al.* (1994)
Crop to pasture	USA	Paired-site	Robles & Burke (1998)
Forest to pasture, pasture to plantation	New Zealand	Paired-site	Ross *et al.* (1999)
Crop to plantation, forest to plantation	USA	Chronosequence	Schiffman & Johnson (1989)
Pasture to plantation	New Zealand	Paired-site	Scott *et al.* (1999)
Forest to crop	Australia	Paired-site	Skjemstad *et al.* (1999)
Pasture to crop	New Zealand	Paired-site	Sparling *et al.* (1992)
Pasture to crop	Canada	Paired-site	Tiessen *et al.* (1982)
Forest to pasture	Brazil	Paired-site	Trumbore *et al.* (1995)
Forest to plantation	Australia	Paired-site	Turner & Kelly (1977)
Forest to plantation	Australia	Paired-site	Turner & Kelly (1985)
Forest to plantation	Australia	Paired-site	Turner & Lambert (1988)
Pasture to crop, crop to pasture	USA	Paired-site	Unger (2001)
Pasture to crop	Australia	Paired-site	Webb & Dowling (1990)
Pasture to crop	Australia	Paired-site	Whitbread *et al.* (1998)
Pasture to plantation	New Zealand	Repeated-sampling	Yeates *et al.* (2000)
Crop to plantation	USA	Chronosequence	Zou & Bashkin (1998)