Above-ground carbon sequestration by early-successional woody vegetation

A preliminary analysis

SCIENCE FOR CONSERVATION 297

Department of Conservation
Te Papa Atawhai

Above-ground carbon sequestration by early-successional woody vegetation

A preliminary analysis

F.E. Carswell; L.E. Burrows and N.W.H. Mason

Cover: Duneland forest, Pouto, Northland.
Photo: Peter Anderson.

CONTENTS

Above-ground carbon sequestration by early-successional woody vegetation

A preliminary analysis

F.E. Carswell; L.E. Burrows and N.W.H. Mason

Landcare Research, PO Box 40, Lincoln 7640, New Zealand
Email: carswellf@landcareresearch.co.nz

ABSTRACT

New Zealand's ratification of the Kyoto Protocol offers considerable opportunity for the sale of 'carbon credits' from newly created forests. Given that forests need to have arisen since 1989 to be eligible for carbon credits, shrublands currently represent the biggest potential for Kyoto-compliant carbon (C) gain on conservation land. Little published information currently exists on the potential annual gain of C (sequestration) in shrublands, or whether significant differences occur between different shrubland types. In this study, change in above-ground C stock was measured over an approximate 5-year period and consequent rates of C sequestration were calculated for six exotic seral and six mānuka (*Leptospermum scoparium*) shrubland plots. Mean C stocks were 73 ± 13 t CO_2/ha for exotic shrubland plots and 76 ± 33 t CO_2/ha for mānuka plots. Mean sequestration rates were 2.7 ± 1.5 t CO_2/ha/y and 2.0 ± 1.0 t CO_2/ha/y for exotic and mānuka shrublands respectively. Given the wide range in transition points between shrub and forest stages of succession with respect to C gain estimated in this preliminary study, we recommend more detailed investigation of long-term consequences of C storage during secondary succession. We also note that widely divergent estimates of sequestration rate could be derived depending on the time interval between measurements of stock change. We tested a simplified shrubland measurement methodology and found that predicted C stocks did not differ significantly from those estimated using the standard method.

Keywords: carbon, exotic seral vegetation, mānuka, sequestration, shrubland, succession

1. Introduction

New Zealand's ratification of the Kyoto Protocol binds the country to take responsibility for gross greenhouse gas emissions greater than those produced in 1990. Through adoption of Article 3.3 of the Protocol, New Zealand may now generate 'emission removal units' (commonly referred to as 'carbon credits') through the establishment of new forests on land that was not forested at 31 December 1989. Kyoto Protocol units associated with these forests can be used to offset national gross greenhouse gas emissions, or can be sold to other countries with Kyoto Protocol liabilities, thereby conferring substantial monetary value. Conversely, a reduction in C stocks could potentially be an additional source of emissions and, therefore, a major financial liability. Current indications suggest that New Zealand may be a net seller of units at the end of the Kyoto Protocol first commitment period[1] (MfE 2009)—the need for accurate estimation of the units accruing to new forests is now urgent. Given that the Department of Conservation (DOC) is responsible for the single biggest biomass carbon (C) stock under one managing authority or owner in New Zealand, DOC will make a major contribution both to new forests as C sinks and the protection and enhancement of C sinks in existing forests.

DOC administers c. 30% of New Zealand's land area and c. 60% of this conservation land is currently in forest or forest–shrubland/grassland mixtures (Carswell et al. 2008). Changes in natural processes and/or human-related impacts and management activities in these ecosystems could result in significant changes in their C stocks. The total estimated terrestrial C stock on conservation land is 2396 Mt (or 8785 Mt CO_2)[2], across five pools (i.e. above- and below-ground vegetation, litter, coarse woody debris and soil C; Carswell et al. 2008). About 80% of the existing stock on conservation land is in indigenous forest or forest–shrubland vegetation classes, and about 50% is estimated to reside in the soil pool. A small change in the per-hectare C stocks over a large area would result in a large increase or decrease in the total C stocks relative to New Zealand's emissions of greenhouse gases. A large biomass change in land-cover from grassland to forest over a relatively small area would also result in considerable additional biomass C stocks and sequester[3] C over a long period of time.

Information on the amount of C stored in broad classes of New Zealand vegetation is currently available (e.g. Tate et al. 1997; Hall et al. 2001), but little is known about sequestration rates, particularly for the initial stages of secondary succession to forest. In this study we focus on one type of secondary succession, namely the transition from exotic grassland to continuous woody

---

[1]   Generally referred to as CP1, the first commitment period of the Kyoto Protocol refers to the time period (2008–2012) over which a country is required to reduce its emissions to the level agreed to during negotiations with all Kyoto signatories. New Zealand has committed to reducing its overall emissions to those of 1990.

[2]   Here, we use CO_2 to denote the number of tonnes (t) of CO_2 that have been effectively removed from the atmosphere and stored as carbon in the soil and/or vegetation. Each tonne of stored C is equivalent to 3.67 t of CO_2 removed from the atmosphere.

[3]   The process of removing C from the atmosphere and storing it in vegetation is commonly referred to as sequestration. Actively growing trees are therefore said to sequester C in woody biomass.

cover. Rates of exotic pasture invasion by woody species are varied and appear to depend on a number of factors, including presence of a woody species that is sufficiently competitive to invade exotic pasture. Exotic species, such as gorse (*Ulex europaeus*) and buddleia (*Buddleja davidii*), are thought to have displaced mānuka (*Leptospermum scoparium*) and kānuka (*Kunzea ericoides*) in many lowland landscapes and can now dominate early forest succession (Smale 1990; Sullivan et al. 2007), although large site-to-site variation has been observed. It is expected that in most instances exotic woody cover will progress to indigenous forest (e.g. Williams 1983; Wilson 1994), but long-term successional trajectories may differ from those occurring under mānuka–kānuka successions (Sullivan et al. 2007). The short-term consequences on biomass accumulation of various transitions between woody vegetation types are almost entirely unknown for New Zealand shrublands and forests.

The purpose of this study was to determine early rates of C sequestration in common shrub types on mesic sites that are likely to proceed to tall forest. We remeasured twelve permanent plots selected from a range of datasets and ascribed each plot to one of two broad shrubland classes—namely those dominated by an indigenous shrubland species, i.e. mānuka, or those dominated by exotic seral species (e.g. gorse, Scotch broom (*Cytisus scoparius*) or buddleia). The study is preliminary in nature because of the small number of plots studied. Remeasurement allowed calculation of stocks of above-ground live C and sequestration rates for each plot and across each broad shrubland type.

2. Objectives

In order to address some gaps in existing knowledge of early succession C sequestration rates this study aimed to:

- Measure above-ground live C stocks in, respectively, a sample of seral exotic shrublands and a sample of indigenous (mānuka) shrublands.

- Measure the rate of above-ground live C sequestration in the samples of regenerating exotic seral and mānuka shrublands.

- Compare estimates of above-ground live C stocks derived using LUCAS[4] shrubland methodology with those derived from another (the 'quarters') methodology for our sample plots.

- Document, where possible, changes in live C storage during transition from exotic seral and mānuka shrublands to later-successional forest vegetation.

---

[4] Land Use and Carbon Analysis System; see section 3.1.

3. Methods

Stocks of above-ground live C and rates of sequestration in shrublands dominated by either exotic seral species or mānuka were investigated using repeat measurements of permanent plots. Plots were subjectively chosen as those likely to proceed to tall forest; i.e. located in places with moderate moisture, altitude, slope (Table 1) and local seed source. These two broad shrubland classes will be highly suitable for minimum-interference management to promote new forests. Two methods were used and compared to collect shrub data. Shrublands in the process of transition from seral species to later-successional forest species have been specifically signalled as such in the discussion of the magnitude and size of the C stock change.

3.1 DATA COLLECTION AND C STOCK CALCULATION

Carbon sequestration was estimated from remeasurement of 12 permanent plots (20 × 20 m) that had been previously measured using either the Land Use and Carbon Analysis System (LUCAS) shrubland methodology (Payton et al. 2004), or a method closely approximating it (e.g. Coomes et al. 2002; Richardson et al. 2004). Six plots in each of exotic seral and mānuka shrubland were selected across three databases. Remeasured plots came from either Landcare Research's EBEX21® database (Carswell et al. 2003), the Ministry for the Environment's (MfE's) South Island transect data (Coomes et al. 2002) or the LUCAS indigenous forest and shrubland plot network. Plots were remeasured in order to estimate changes in C stocks over a minimum of 4.5 years. Exotic seral plots were located in the central North Island (4 plots), near Christchurch (1) and near Dunedin (1). Mānuka shrubland plots were located in the central North Island (1), West Coast of the South Island (2), North Canterbury (2) and in the Marlborough Sounds (1).

The methodology used to collect and analyse data was that of LUCAS and a full description of methods is provided in Coomes et al. (2002), Payton et al. (2004) and Peltzer & Payton (2006), and we therefore only briefly describe methods here. Initial data from each source were collected as follows: EBEX plot data were

TABLE 1. AVERAGE SITE CONDITIONS FOR SIX EXOTIC SERAL AND SIX MĀNUKA SHRUBLAND PLOTS REMEASURED IN THE CURRENT STUDY. ALL VALUES ARE SHOWN ± ONE STANDARD ERROR.

SITE FACTOR	EXOTIC SERAL	MĀNUKA
Elevation (m a.s.l.)	290 ± 35	414 ± 107
Slope (°)	11 ± 5	21 ± 7
Rainfall (mm)	1174 ± 91	2018 ± 315
Annual water deficit (mm)*	40 ± 24	1 ± 1
Mean annual temperature (°)	11.4 ± 0.5	10.2 ± 0.6
Cation exchange capacity (cmol/kg)	11 ± 2	22 ± 7
Phosphorus retention (%)	39 ± 7	38 ± 5

* This is a measure of the excess of evaporation over rainfall. Where rainfall exceeds evaporation, the annual water deficit is expressed as 0 mm (Leathwick et al. 2002). National mean annual water deficit is 36.2 mm and the range is 0–396 mm.

collected in March 2002, South Island transect data were collected in January 1999, and LUCAS plot data were collected between May 2002 and January 2004. All plots were remeasured in August–September 2008.

Above-ground C was assessed through measurement of both the live tree (≥ 2.5 cm diameter at breast height, dbh) and shrub components. Shrub cover was assessed as either continuous or in discrete clumps before measurement, and only one method was applied to an entire plot.

3.1.1 Estimating C of continuous shrub cover

Continuous shrub cover is mingled woody vegetation above 30 cm in height where individual plants or clumps are impossible to separate within a 20 × 20 m plot. The volume of shrubs (assuming cuboid geometry) was calculated from mean height and cover of each species within 5 × 5 m sub-plots. These were summed for the entire 20 × 20 m plot and converted to above-ground biomass using a generic formula of:

DM = V × dens.

where *DM* = dry mass, *V* = volume and dens. = shrub density (Coomes et al. 2002).

With the exception of mānuka, values for shrub densities used were those calculated from destructive harvests during year 1 of LUCAS data collection, and tabulated in Peltzer & Payton (2006). A shrub density value of 1.54 kg/m^3 (Coomes et al. 2002) was used for mānuka, as this was considered a more representative value for this species. Where a species was not represented by the harvested data presented in Peltzer & Payton (2006), densities of other species in the same genus were substituted, or else the median value across all shrub types was used. Note that the mean value was not considered suitable because it is easily over-inflated by a small number of very high density values, unlike the median (Lieberman et al. 1985; Laurance et al. 2004).

Biomass was summed across the species present to give the total shrub dry mass, and 50% of this biomass was assumed to be C (Coomes et al. 2002).

3.1.2 Estimating C of discrete shrubs

Discrete shrub cover was defined as woody vegetation above 30 cm in height where individual shrubs or clumps of shrubs could be identified within a 5 × 5 m sub-plot. These could potentially comprise several species. For each shrub or shrub clump with at least 50% of its canopy occurring within the plot, the maximum height, maximum canopy diameter, and the diameter perpendicular to maximum diameter were recorded, to allow estimation of the shrub volume as an ellipse (Carswell et al. 2001). Volume was summed for each species within the 20 × 20 m plot. Volumes were converted to biomass using previously published allometric relationships[5] for two species (namely, Scotch broom (Carswell et al.

---

[5] Allometric relationships are simple formulae that convert the plant attributes that are easily measured in the field to attributes that are more difficult to measure. For example, plant mass can be reliably calculated from measured values of tree diameter and height. In the current study, the 3-dimensional space occupied by shrubs was measured and converted to shrub biomass on the basis of pre-determined relationships.

2001) and tauhinu (Carswell et al. 2006)), an unpublished allometric relationship for bracken (S.J. Richardson, pers. comm.) and the generic formula of Coomes et al. (2002) given above, for all other species.

Biomass was summed across the species present to give the total discrete shrub biomass, and 50% of this biomass was assumed to be C.

3.1.3   Estimating C in live trees

Total above-ground biomass (AGB) was calculated using the following formula (Coomes et al. 2002):

$$AGB = 0.0000598 \times \rho \times (dbh^2 \times ht)^{0.946} \times (1 - 0.0019 \times dbh) + (0.03 \times dbh^{2.33}) + (0.0406 \times dbh^{1.53})$$

where ρ = wood density (kg/m^3), dbh = diameter at breast height (cm), ht = tree height (m). Wood density values were obtained from Peltzer & Payton (2006). Where species-specific values were unavailable, values for other species in the same genus were used. Finally, tree biomass was assumed to comprise 50% of C (Coomes et al. 2002).

Total C contents of plots were calculated by adding C in live trees to that in either discrete or continuous shrubs. Total values were corrected for plot slope by dividing the C stock estimate by the cosine of the slope angle. C values were then converted to CO_2 using a multiplier of 3.67 and expressed as t CO_2/ha.

Sequestration rates were calculated as the difference in C stocks between successive measurements and estimated as t/ha/y. For some plots, independent verification of successional age was also available, enabling calculation of mean annual sequestration rate over the life of the current succession.

3.1.4   A test of an alternative method for assessing shrub volume

An alternative 'quarters' method was also used to estimate shrub cover on 11 of the plots in order to determine whether a rapid assessment of this component is possible. The alternative methodology involves dividing the plot into four 10 × 10 m quarters and assessing all woody cover ≤ 2.5 cm dbh as the total area of the quarter covered per species. Area occupied by each species was assessed through use of the sub-dividing plot tapes. Height was measured at ten evenly spaced points within each discrete shrub clump and a mean height given for that clump. If the sub-plot was covered with continuous shrub, height was measured at ten evenly spaced points within each species and a mean height calculated per species. The same relationships as those given earlier were used to convert these measurements to volume and C. Estimates of above-ground live C stocks were compared with those obtained through use of the LUCAS methodology as described above.

3.2   STATISTICAL ANALYSIS

A repeated-measures analysis of variance (ANOVA) was used to test for differences between measurements using GenStat 6th edition. Differences in sequestration rates between exotic and indigenous seral shrubland were investigated using ANOVA, also in GenStat. Estimates of C per plot obtained using the LUCAS methods, as opposed to the quarters method, were compared using a paired t-test in R version 2.7.2. (R Development Core, www.r-project.org/).

4. Results

Above-ground C stocks generally increased between measurements in both exotic seral and mānuka shrubland plots (Fig. 1). Remeasured plots had sequestered an additional 37% of CO_2, on average, compared with the initial measurement ($F = 6.72$, df = 1, 11, $P = 0.025$). Combining data from all shrubland plots at the time of remeasurement gave an average above-ground C stock of 74 ± 17 t CO_2/ha (1 SE given). Mean above-ground C stocks were 73 ± 13 t CO_2/ha for exotic shrubland plots and 76 ± 33 t CO_2/ha for mānuka plots. Above-ground C stocks were more variable for mānuka plots, reflecting the wide range of successional states sampled—from recent colonisation of former pasture lands (low C stocks) to near-forest (high C stocks).

Figure 1. Above-ground C stock of 12 shrubland plots at the time of remeasurement, plotted as a function of above-ground C stock at the time of initial plot measurement. Plots are from either exotic or mānuka shrubland, and are depicted according to whether the plots were in early, later or transitional seral stages. Also shown is the 1:1 line representing no change in C stock.
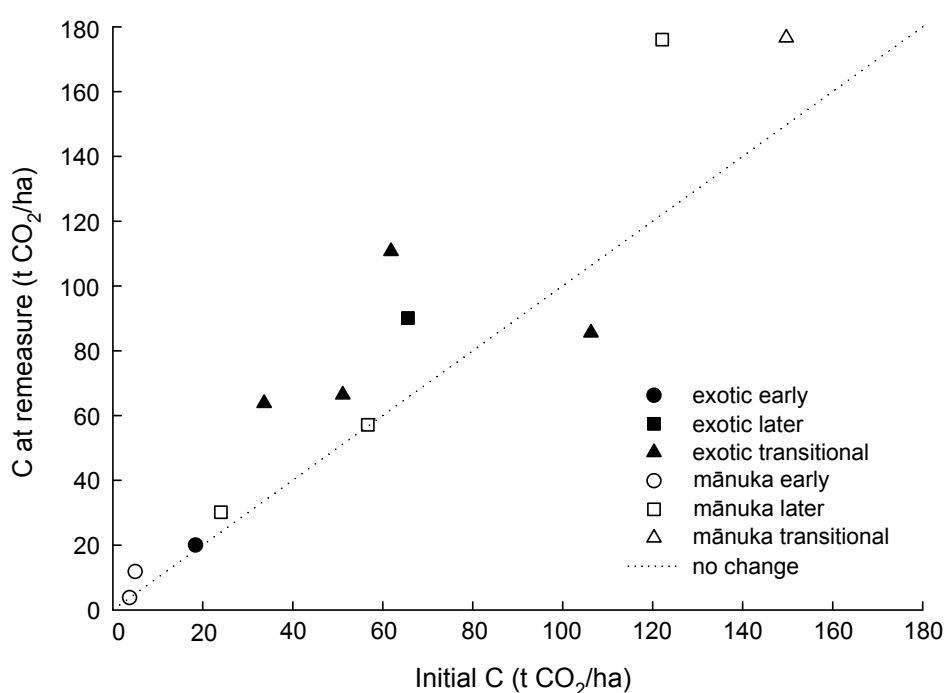
Sequestration rates did not differ statistically between shrubland types ($F = 0.14$, df = 1, 11, $P = 0.7$), although a single negative value in the exotic shrubland group is likely depressing the rate for that group as a whole (Fig. 2). Mean sequestration rate was 2.3 ± 0.9 t CO_2/ha/y overall. Indicative means for each type of shrubland were 2.7 ± 1.5 t CO_2/ha/y and 2.0 ± 1.0 t CO_2/ha/y for exotic and mānuka shrublands, respectively.

Because the ages of the shrubland were known for some of the plots, it was possible to derive a mean annual sequestration rate based on successional age for those plots only. In these instances the rate was calculated as current above-ground live C divided by the successional age. We assumed that all above-ground C stock was additional to what would have been present on plots at the initiation of succession. Given aerial imagery verification that the former land cover was bare ground for all except one of the exotic seral plots located on a riverbed (i.e. the vegetation is constantly renewed by successive floods), this seems a reasonable assumption. Results from this calculation show all sites had a positive accumulation in above-ground C stock (Table 2) over the longer time period.

Values for shrub C stock did not differ significantly between plots ($t = 1.2971$, df = 10, $P = 0.22$) when using the quarters shrubland methodology, as opposed to the standard LUCAS methodology. Figure 3 shows an approximately equal distribution of points above and below the 1:1 line that represents an exact match between C stock measurements obtained by each method, i.e. C is not systematically over- or under-predicted using the quarters method, compared with the LUCAS method.

Figure 2. Sequestration rates of both exotic and mānuka shrubland plots as a function of initial above-ground C stock. Plots are depicted according to whether the plots were in early, later or transitional seral stages.
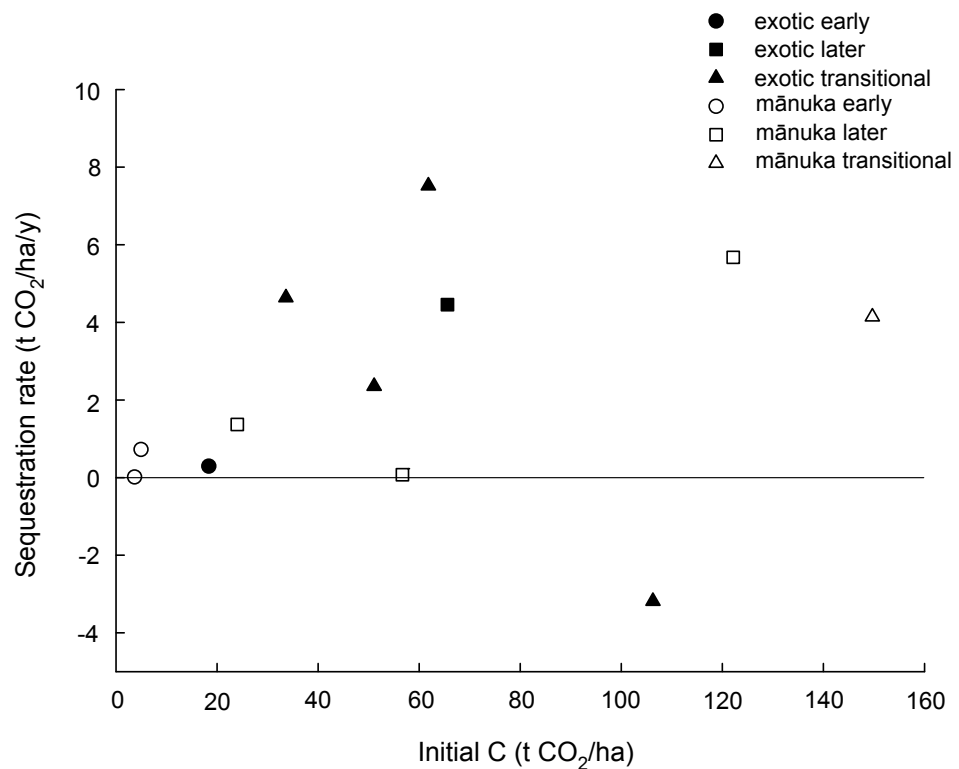

TABLE 2.   SEQUESTRATION RATE OVER DURATION OF SUCCESSION TO DATE FOR FIVE PLOTS FOR WHICH SUCCESSIONAL AGE IS KNOWN.

PLOT	AGE (YEARS)	SEQUESTRATION RATE (t CO_2/ha/y)
Exotic 1	18	6.2
Exotic 3	18	3.5
Exotic 4*	5	3.7
Exotic 5	18	4.0
Exotic 6	18	4.8

*   Note that although plot E4 is only 5 years old, vegetation is constantly being renewed (riverbed) so there will be no net sequestration unless the site stabilises such that the succession can proceed.

Figure 3. Estimated above-ground C stock per plot using the alternative 'quarters' methodology for shrubs only and plotted as a function of above-ground C stock estimated through use of LUCAS methodology. Also shown is the 1:1 line that would represent a perfect match between estimates.
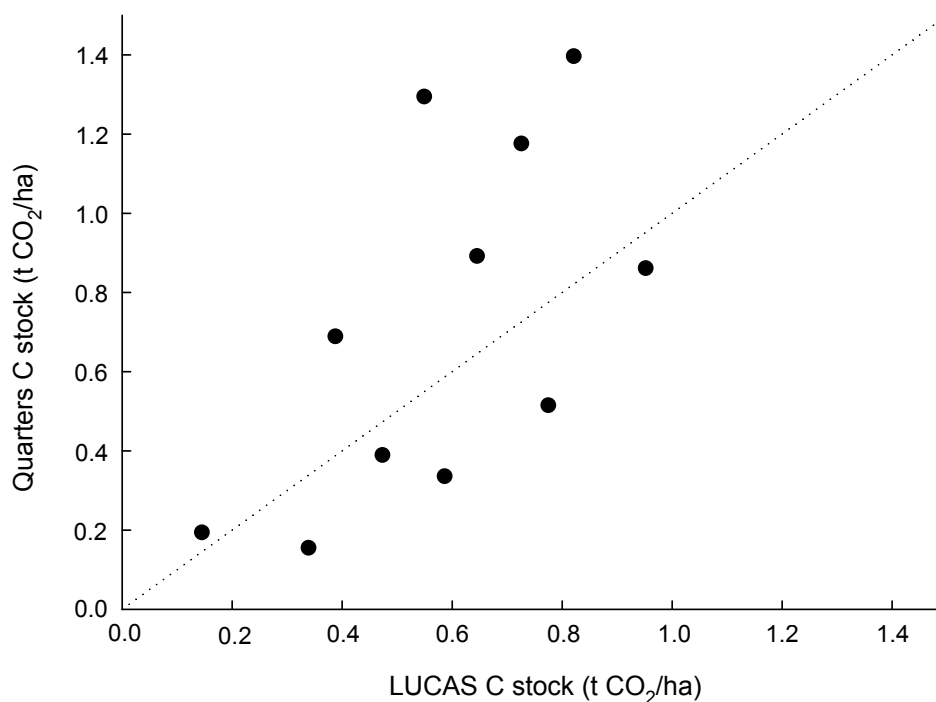
5. Discussion

The mean estimate of above-ground C stocks in this study (74 t CO_2/ha) was similar to a previous mean estimate of 78 t CO_2/ha reported for shrubland plots measured on another South Island transect (Coomes et al. 2002). It is possible to estimate CO_2/ha for total live woody biomass by adding a component for below-ground biomass (roots) to the above-ground estimate. It is widely accepted that below-ground biomass can be estimated at 25% of above-ground biomass (Phillips & Watson 1994; Cairns et al. 1997; Coomes et al. 2002). Adding such a component to our estimate gave a value of 92.5 t CO_2/ha for total live woody biomass, compared with a value of 98 t CO_2/ha from Coomes et al. (2002). Values from both studies were higher than those obtained for shrubland plots using the LUCAS data collection. Total live biomass on those plots (excluding subalpine shrublands) was estimated at 52 t CO_2/ha (Beets et al. 2009). Given that LUCAS has been designed to give an unbiased estimate of forest and shrubland C for the whole country and across the full range of shrubland types, we would expect the mean values derived from LUCAS to be more truly reflective of average C stocks for shrubland. Our plots merely represent a small subset of the possible range of New Zealand shrublands, and they were subjectively chosen (as plots likely to proceed to tall indigenous forest), so the variation between our estimated averages and those from LUCAS should not be surprising.

The above-ground C stocks estimated for the mānuka-dominated plots we sampled spanned a range of values. Above-ground C in the top end of the range was 180 t CO_2/ha, which is very close to the 183 t CO_2/ha C obtained for live above-ground mānuka stands at maximum biomass according to modelled trajectories

of biomass accumulation during natural succession (Hall 2001). A previous study of biomass accumulation in regenerating mānuka measured 103 t CO_2/ha of above-ground C storage at only 10 years in age (Walcroft et al. 2002). When only the live components of above-ground biomass in stands dominated by gorse were included and converted to C stocks, the estimates provided by Egunjobi (1971) suggested that 104 t CO_2/ha were stored in a 7.5-year-old stand. The highest exotic-dominated shrubland in the current study (111 t CO_2/ha) was for a stand dominated by buddleia. This value was comparable to that estimated by Egunjobi (1971), although ages for the two sites are different (we estimate mean age to be around 18 years for the buddleia-dominated stand measured in our study).

Calculated sequestration rates for the c. 5-year measurement interval[6] in the current study were similar to, if not a little below, the c. 3 t CO_2/ha/y that can be calculated on the basis of mean total live C in tall forests (594 t CO_2/ha; Hall et al. 2001) divided by the c. 200 years to first attain tall forest status (Hall 2001). Mean mānuka sequestration rates in this study were significantly lower than those reported for early-growth (first 40 years) mānuka–kānuka shrubland, which can sequester 7–9 t CO_2/ha/y, with variation largely ascribed to site fertility and water availability (Trotter et al. 2005). However, the mānuka site with the highest sequestration rate in the current study was storing c. 6 t CO_2/ha/y, akin to that observed by Trotter et al. (2005). We acknowledge that observed rates of sequestration could be more strongly driven by environmental conditions than species differences, as in the study of Trotter et al. (2005). However, given the small number of plots in the current study we were unable to partition these differences from those relating to shrubland type. Further study of the drivers of sequestration rate is warranted.

The large variation observed in the current study reflects the wide variety of successional stages sampled, and the variable nature of random samples. The sites exhibiting low rates of sequestration were generally those in the earlier stages of grassland colonisation, and were not yet fully covered by woody species. The one plot that lost live above-ground C during the 5-y measurement interval was in transition between a buddleia-dominated seral stage and forest. The observed loss is not unexpected as at these transitional stages, forest biomass (and consequently C) accumulation can either plateau or decline (Peet 1981). The plot consisted largely of senescing or dead buddleia stems (excluded from our measurement of live above-ground biomass) and a dense understorey of small (<5 cm dbh) *Coprosma robusta* and *Melicytus ramiflorus* trees. While the net change in C over the measurement interval was negative, the overall gain in C since the initiation of the succession was positive, and occurred at a rate of 4.8 t CO_2/ha/y when averaged over the 'lifetime' of the current forest succession. It is of note, however, that the highest rate of sequestration observed for mānuka in the current study was also in a stand undergoing transition to a later-successional species, in this case *Nothofagus solandri* var. *cliffortioides* (mountain beech). The positive rate of sequestration in this case could be an artefact of the measurement interval, i.e. perhaps the senescent phase was

---

[6] Remeasurement interval did not systematically bias the estimated sequestration rate—the two plots with a 9.5-y measurement interval showed both the second highest and third lowest rates of sequestration. All other plots fell within a 2-y remeasurement span (4.5–6.5 y) that did not correlate with estimated sequestration rate.

missed, or, as seems more likely from observation of the stand, the replacement of mānuka by beech occurred rapidly, with any loss in mānuka biomass being more than compensated for by beech growth. It would be helpful to know if it is possible to design shrubland successions that predominantly exhibit this pattern of biomass accumulation, so that owners of regenerating indigenous forests will not incur a financial liability in successions where the senescent phase results in significantly negative sequestration, should they choose to engage with C trading markets[7].

Estimates for C stocks at the plot scale were comparable when shrubland cover was measured using either the quarters or the LUCAS methodology, although when the quarters method predicted a higher C stock than the LUCAS method, the differences appeared larger than when it predicted a smaller C stock (Fig. 3). The plots that diverged widely in the estimates were generally those where the dominant shrub cover was above 2.5 m in height. This is thought to be a function of the difficulty of estimating overhead proportional cover over a large (10 × 10 m) area. Accuracy on a 10 × 10 m sampling scale was definitely better in shorter shrubland, where the whole plot could be viewed from fewer vantage points. A more intensive investigation over a larger number of plots would be helpful for examining the best ways of applying the quarters approach.

We have presented a range of shrubland C stocks and sequestration rates as an indication of what could be expected on conservation land, should more grassland be allowed to revert to indigenous forest using natural succession. The data presented here are in no way intended to represent the national potential for shrubland C sequestration and our choice of sample sites was greatly restricted by the following requirements: (a) that plots were representative of 'typical' successions that might be utilised to increase C stocks on conservation land; (b) that plots were permanent and had been previously sampled for C stocks using the standard LUCAS measurement methodology; (c) that time elapsed since the last measurement was at least 4.5 years so that changes in stocks would be greater than measurement errors; (d) that plots were both capable of succeeding to tall forest and occupied moderate environmental conditions; and (e) that plots were accessible by means other than helicopter in order to reduce operational costs. What is now required to build predictive capacity is a systematic sampling strategy across a wide variety of environments, dominant shrubland species and successional ages. This would also enable quantification and explanation of variation through both time and space. The planned remeasurement of all the LUCAS plots will be a helpful step in this process.

The calculation of C stock for New Zealand shrubland plots remains problematic because of a lack of underlying empirical data and relationships concerning species-specific allometries for volume and also shrub densities. We do not have robust allometric relationships for *Dicksonia* spp. or *Cyathea* spp. (tree ferns), nor for commonly occurring monocotyledons such as *Cordyline australis* (cabbage trees) and *Freycinetia banksii* (kiekie). Other non-woody species such as *Cortaderia* spp. or *Carex* spp. can contribute significant biomass to a plot

---

7   Current domestic policy allows landowners with post-1989 forests to gain 'carbon credits' that they may exchange for money by participating in C trading markets as the forests grow, but they must pay back credits when C is lost between successive measurements (c. 5-yearly).

but no allometric functions currently account for these. At the present time, C in tree ferns and cabbage trees is still calculated using an allometric relationship that assumes increasing branch biomass with increasing stem diameter. This is clearly an invalid assumption. A promising alternative for tree ferns exists (Beets et al. 2008), but as yet is underpinned by too small a sample to be convincing. Shrub density data is still missing for some relatively commonly occurring early successional species. In addition, the systematic survey of wood density change with environmental factors such as site fertility, rainfall, altitude and shrub maturity is suggested.

Despite the apparent similarity in C stocks and rates of sequestration by exotic-seral and mānuka shrublands, early-successional species do not appear equivalent in their influence on the species that will succeed them (e.g. Lusk & Ogden 1992; Meurk & Hall 2006; Sullivan et al. 2007). This suggests that the long-term consequences for C accumulation resulting from promotion of particular successions are complex. Biomass accumulation during forest succession can follow a number of trajectories (e.g. Peet 1981). Some of these have been modelled for New Zealand forests using LINKNZ, a generalised temperate forest gap model (Hall & Hollinger 2000). However, data on either biomass or C accumulation during NZ forest successions are very scarce. Existing data largely come from beech forests (e.g. Schöenenberger 1984; Davis et al. 2003) and exclude the early stages of secondary succession. On the other hand, some data are available that show only C sequestration for the early successional stages (e.g. Scott et al. 2000; Trotter et al. 2005).

Potential management actions during successions to enable maximised C storage in forest ecosystems are even less clear. For example, does the greatest potential for manipulation of a desirable successional trajectory exist at the early stages of succession (i.e. through very early introduction of desirable later-successional species), or during the actual transition of the seral shrub species to tall forest species? Successional trajectories with little, or no, loss of C during transition to tall forest species would be ideal for minimum-interference management (e.g. timely manipulation of grazing herbivore density (Burrows et al. 2008)). For successions expected to experience C loss, i.e. those experiencing tightly synchronised senescence in the seral species, the period and the total amount of C lost during this transition (including all pools) needs to be quantified such that cost-benefit analysis can be performed for more intensive management actions that might minimise C loss at the time of transition, e.g. through early supplementary planting of high biomass species.

Given the significant opportunity for land managers to contribute to New Zealand's efforts to reduce net greenhouse gas emissions, an urgent need exists for knowledge of changes in C storage during forest succession and the early species compositions and transitions that could be manipulated to minimise C losses and maximise long-term C storage gains.

6. Conclusions

Post-1989 naturally regenerating shrublands offer significant potential to gain C revenue without excessive capital expenditure. Conservation lands contain the largest pool of such shrublands under single management and have much to contribute to national accounts. We measured positive rates of mean C sequestration over a c. 5-year timespan for both exotic seral and mānuka shrublands and found that:

- Mean C stocks were 73 ± 13 t CO_2/ha for exotic seral shrubland plots and 76 ± 33 t CO_2/ha for mānuka plots.

- Mean sequestration rates were 2.7 ± 1.5 t CO_2/ha/y for exotic seral shrublands and 2.0 ± 1.0 t CO_2/ha/y for mānuka shrublands.

- A simpler, and more quickly applied, field method for estimation of shrubland cover (the 'quarters' method) provided C estimates of similar magnitude to those derived from LUCAS methodology.

- Transitions between early-successional shrubland species (such as buddleia and mānuka) and later-successional broadleaved species may, or may not, result in short-term loss of above-ground C stocks. The effect appears to be site- and species-dependent, but more data and an improved theoretical understanding of the processes controlling transitions between New Zealand seral stands are required if we are to understand the consequences of these transitions as they relate to C, and thus have any possibility of managing them.

7. Recommendations

Uncertainties around the time-span and magnitude of net sequestration of C by regenerating shrublands still need to be resolved to enable management of natural forests for C gain. Priorities for resolution are given in the following recommendations:

- Changes in C storage during key New Zealand secondary successions from pasture to tall forest should be quantified. A useful first step would be to utilise remeasured plots in the National Vegetation Survey (NVS) database that fulfil specific criteria. Transitions of mānuka-kānuka to podocarp-hardwood and/or beech forest, and the successions to lowland coastal forest, would be particularly relevant to land managers wishing to maximise C gain in new forests.

- Robust allometric functions should be derived for key shrubland species that have not yet been investigated, such as *Coprosma robusta*, *Melicytus ramiflorus*, *Pseudopanax arboreus*, *Cordyline australis*, tree ferns by species and a generic 'woody coprosma' function.

- Research should be initiated to determine whether some successions suffer little C loss during the senescent phase and transition to later-successional species.

8.   Acknowledgements

9.   References

Beets, P.N.; Oliver, G.R.; Kimberley, M.O.; Pearce, S.H. 2008: Allometric functions for estimating above ground carbon in native forest trees, shrubs and tree ferns. Scion contract report (No. 10048) prepared for the Ministry for the Environment, Wellington.

Beets, P.N.; Kimberley, M.O.; Goulding, C.J.; Garrett, L.G.; Oliver, G.R.; Paul, T.S.H. 2009: Natural forest plot data analysis: carbon stock analyses and re-measurement strategy. Scion contract report prepared for the Ministry for the Environment, Wellington.

Burrows, L.E.; Peltzer, D.A.; Bellingham, P.J.; Allen, R.B. 2008: Effects of the control of introduced wild animal herbivores on carbon stocks. Landcare Research Contract Report LC0708/087 prepared for the Department of Conservation. 68 p.

Cairns, M.A.; Brown, S.; Helmer, E.H.; Baumgardner, G.A. 1997: Root biomass allocation in the world's upland forests. *Oecologia 111*: 1–11.

Carswell, F.; Brignall-Theyer, M.; Burrows, L.; McKenzie, S. 2006: An allometric relationship between shrub volume and carbon for tauhinu (*Ozothamnus leptophyllus*). LC0506/146. 14 p.

Carswell, F.E.; Frame, B.; Martin, V.; Turney, I. 2003: Exchanging emissions for biodiversity— in pursuit of an integrated solution in New Zealand. *Ecological Management and Restoration 4*: 85–93.

Carswell, F.E.; Mason, N.W.H.; Davis, M.R.; Briggs, C.W.; Clinton, P.W.; Green, W.; Standish, R.J.; Allen, R.J.; Burrows, L.E. 2008: Synthesis of carbon stock information regarding conservation land. Landcare Research Contract Report LC0708/071 prepared for the Department of Conservation (unpublished). 116 p.

Carswell, F.; Pearce, G.; Burrows, L.; Rogers, G.; Buxton, R.; Dangeron, F. 2001: Allometric functions for shrubs. Task B2 for the Carbon Monitoring System. Joint Landcare Research and Forest Research contract report JNT0001/133. 14 p.

Coomes, D.A.; Allen, R.B.; Scott, N.A.; Goulding, C.; Beets P. 2002: Designing systems to monitor carbon stocks in forests and shrublands. *Forest Ecology and Management 164*: 89–108.

Davis, M.R.; Allen, R.B.; Clinton, P.W. 2003: Carbon storage along a stand development sequence in a New Zealand *Nothofagus* forest. *Forest Ecology and Management 177*: 313–321.

Egunjobi, J.K. 1971: Ecosystem processes in a stand of *Ulex europaeus* L. I. Dry matter production, litter fall and efficiency of solar energy utilization. *Journal of Ecology 59*: 31–38.

Hall, G.M.J. 2001: Mitigating an organisation's carbon emissions by native forest regeneration. *Ecological Applications 11*: 1622–1633.

Hall, G.M.J.; Hollinger, D.Y. 2000: Simulating New Zealand forest dynamics with a generalized temperate forest gap model. *Ecological Applications 10*: 115–130.

Hall, G.M.J.; Wiser, S.K.; Allen, R.B.; Beets, P.N.; Goulding, C.J. 2001: Strategies to estimate national forest carbon stocks from inventory data: the 1990 New Zealand baseline. *Global Change Biology 7*: 389–403.

Laurance, W.F.; Nascimentoa, H.E.M.; Laurance, S.G.; Condit, R.; D'Angelo, S.; Andrade, A. 2004: Inferred longevity of Amazonian rainforest trees based on a long-term demographic study. *Forest Ecology and Management 190*: 131–143.

Leathwick, J.; Morgan, F.; Wilson, G.; Rutledge, D.; McLeod, M.; Johnston, K. 2002: Land Environments of New Zealand: a technical guide. Ministry for the Environment, Wellington. 237 p.

Lieberman, D.; Lieberman, M.; Hartshorn, G.S.; Peralta, R. 1985: Growth rates and age-size relationships of tropical wet forest trees in Costa Rica. *Journal of Tropical Ecology 1*: 97–109.

Lusk, C.; Ogden, J. 1992: Age structure and dynamics of a podocarp-broadleaf forest in Tongariro National Park, New Zealand. *Journal of Ecology 80*: 379–393.

Meurk, C.D.; Hall, G.M.J. 2006: Options for enhancing forest biodiversity across New Zealand's managed landscapes based on ecosystem modelling and spatial design. *New Zealand Journal of Ecology 30*: 131–146.

MfE (Ministry for the Environment) 2009: Net Position Report 2009. Ministry for the Environment, Wellington.

Payton, I.J.; Newell, C.L.; Beets, P.N. 2004: New Zealand Carbon Monitoring System indigenous forest and shrubland data collection manual. Lincoln and Rotorua, Landcare Research and Forest Research report for the Ministry for the Environment. 68 p.

Peet, R.K. 1981: Changes in biomass and production during secondary forest succession. Pp. 324–338 in West, D.C.; Shugart, H.H.; Botkin, D.B. (Eds): Forest succession, concepts and application. Springer-Verlag, New York.

Peltzer, D.A.; Payton, I.J. 2006: Analysis of Carbon Monitoring System data for indigenous forests and shrublands collected in 2002/03. Landcare Research Contract Report LC0506/099 prepared for the Ministry for the Environment (unpublished). 63 p.

Phillips, C.J.; Watson, A.J. 1994: Structural tree root research in New Zealand: a review. *Landcare Research Science Series 7*. Landcare Research, Lincoln. 71 p.

Richardson, S.J.; Burrows, L.E.; Carswell, F.E. 2004: A plot-based method for monitoring C accumulation in EBEX21® vegetation. Landcare Research contract report LC0304/143. Landcare Research, Lincoln. 18 p.

Schöenenberger, W. 1984: Above-ground biomass of mountain beech (*Nothofagus solandri* (Hook.f) Oerst. var. *cliffortidoides* (Hook.f) Poole) in different stand types near timberline in New Zealand. *Forestry 57*: 59–73.

Scott, N.A.; White, J.D.; Townsend, J.A.; Whitehead, D.; Leathwick, J.R.; Hall, G.M.J.; Marden, M.; Rogers, G.N.D.; Watson, J.; Whaley, P.T. 2000: Carbon and nitrogen distribution and accumulation in a New Zealand scrubland ecosystem. *Canadian Journal of Forest Research 30*: 1246–1255.

Smale, M.C. 1990: Ecological role of buddleia (*Buddleja davidii*) in streambeds in Te Urewera National Park. *New Zealand Journal of Ecology 14*: 1–6.

Sullivan, J.S.; Williams, P.A.; Timmins, S.M. 2007: Secondary forest succession differs through naturalized gorse and native kanuka near Wellington and Nelson. *New Zealand Journal of Ecology 31*: 22–38.

Tate, K.R.; Giltrap, D.J.; Claydon, J.J.; Newsome, P.F.; Atkinson, A.E.; Taylor, M.D.; Lee, R. 1997: Organic carbon stocks in New Zealand's terrestrial ecosystems. *Journal of the Royal Society of New Zealand 27*: 315-335.

Trotter, C.; Tate, K.; Scott, N.; Townsend, J.; Wilde, H.; Lambie, S.; Marden, M.; Pinkney, T. 2005: Afforestation/reforestation of New Zealand marginal pasture lands by indigenous shrublands: the potential for Kyoto forest sinks. *Annals of Forest Science 62*: 865-871.

Walcroft, A.; Trotter, C.; Townsend, J.; Rodda, N.; Belliss, S.; Vickers, G.; Aye, T. 2002: Biomass accumulation in young regenerating shrubland ecosystems on marginal pastoral farmland. Landcare Research Contract Report LC0102/121 prepared for the Foundation for Research, Science and Technology (unpublished).

Williams, P.A. 1983: Secondary vegetation succession on the Port Hills Banks Peninsula, Canterbury, New Zealand. *New Zealand Journal of Botany 21*: 237-247.

Wilson, H.D. 1994: Regeneration of native forest on Hinewai Reserve, Banks Peninsula. *New Zealand Journal of Botany 32*: 373-383.

How much carbon does shrub vegetation sequester?

Ratification of the Kyoto Protocol means that New Zealand may now generate 'carbon credits' through the establishment of new forests on land that was not forested at 31 December 1989. The need for accurate estimation of the units accruing to new forests is now urgent. Since new forests start as shrubland/ grassland, this study focused on one type of secondary succession: the transition from exotic grassland to continuous woody cover. The aim of this research was to determine early rates of carbon sequestration in common shrub types on sites that are likely to proceed to tall forest.

New Zealand Government