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Carbon balance implications of land use change from pasture to managed eucalyptus forest in Hawaii

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ABSTRACT
Mitigation of climate change via increased plant productivity and soil carbon (C) sequestration during land use change can be a powerful driver of the net greenhouse gas emissions of a sustainable production system. Yet the net climate change mitigation of managed forests is affected by both tradeoffs between C sequestration and non-renewable C emissions and assessment methodology. As a case study, we measured ecosystem stocks to determine the potential C implications of converting pasture to managed eucalyptus forest and compared them with the eucalyptus production system's non-renewable C emissions. The forest border was chosen as the system boundary and operations spanned from forest establishment activities to harvested wood placed at the forest perimeter. Eucalyptus biomass C was 57.2 ± 4.2 Mg C ha^{-1} and soil C stock (to ~1 m depth) was approximately an order of magnitude greater. By the prevalent method for bulk density-based determination of C stock, conversion of pasture to eucalyptus forest significantly increased soil C stock by 17.5 ± 8.0%. However, no significant change was detected by the equivalent soil mass method, a less prevalent but more accurate approach to detecting differences in soil C stock due to land use or management changes. A 7-year eucalyptus production cycle generated 1.0 Mg C ha^{-1} in non-renewable emissions, which was far exceeded by the tree biomass C. Thus, even without significant soil C sequestration, this system provided a substantial climate change mitigation service by offsetting non-renewable energy use and C emissions associated with wood production, and providing opportunities for biofuel and bioenergy products to displace fossil fuel products.

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Introduction

The search intensifies for methods to reduce carbon dioxide (CO_2) enrichment of the atmosphere as global climate change progresses and the implications for rising greenhouse gas (GHG) concentrations multiply [1]. To this end, bioenergy production within sustainable, regional systems can provide multiple benefits that include renewable biomass generation, low non-renewable emissions and additional potential carbon (C) benefits through sequestration in terrestrial ecosystems [2]. Bioenergy production is considered a C-neutral process – as the initial photosynthetic growth offsets the CO_2 released during the conversion to energy by combustion [3]. The growth of a biofuel feedstock removes CO_2 from the atmosphere, contrary to the burning of fossil fuels, which only contributes CO_2 to the atmosphere [3,4]. Each rotation of renewable energy production provides a climate change mitigation service through avoided fossil fuel consumption. However, land use change and non-renewable emissions associated with transportation and fuel conversion also are part of the production system and may contribute to a reduction in system-level benefits or, in some cases, a net overall GHG emission [4–6].

Terrestrial ecosystems, including vegetation and soils, play a key role in the global C cycle [7] and enhancing C sequestration within these pools may be the most cost-effective option for drawing down substantial amounts of atmospheric CO_2 [8]. Management strategies can augment the natural sink strength, for example forest management for woody bioenergy crops [9] or improved grassland management [10]. Globally, soil contains more C than aboveground vegetation and the atmosphere combined, but is highly sensitive to disturbance [11,12]. Depending on management practices and the net effect of land conversion on belowground soil processes, land use change during the implementation of a renewable fuels system can lead to either positive or negative changes to the soil C pool [9]. Production systems that maximize soil C sequestration through select land use changes, conservation practices during preparation and harvest, and site selection for areas with a high inherent capacity for C storage also maximize the potential for a C credit, or offset, to the overall debt [2, 9,13,14].

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Defining, quantifying and valuating terrestrial C resources remain as challenges to the implementation of C markets globally [1,15] even though many voluntary markets, standards and exchanges have developed, and large-scale trials of payments to farmers for following C-based land management strategies proceed worldwide. In the United States, programs such as LandCarbon led by the US Geological Survey are developing protocols for quantifying national resources. The World Bank backs the BioCarbon Fund, which recently was used with mixed results in a large-scale case study in Kenya for the adoption of sustainable agricultural land management (SALM) (Tennigkeit et al. 2013) [16]. Further, voluntary C markets allow a speculative valuation and inclusion of C resources in economic assessment of bioenergy production systems [17]. "Carbon-neutral" renewable fuel systems have biomass production as the centerpiece, but also hold opportunities to further offset non-renewable GHG emissions through conservation silvicultural and agricultural practices [9]. For example, maximizing soil C sequestration, or the long-term storage of C below ground, with management is one path toward production system sustainability [13].

The conversion of biomass to biofuel by rotationally harvesting managed eucalyptus forests in Hawaii is being considered as a viable replacement for fossil fuel as it may help mitigate climate change [18] and provide a locally produced, sustainable fuel source. While conversion of natural ecosystems would have clear negative impacts for conservation and biodiversity, ranch land and fallow/degraded land abandoned by sugar production are considered suitable sites for managed *Eucalyptus grandis* forests [19] that avoid these costs. On the island of Hawaii, much of the 93,078 ha suitable for growing *E. grandis* as a biofuel feedstock currently is in pasture [20]. Volcanic ash-derived soils are predominant on the arable lands of the island of Hawaii and have a particularly high capacity for soil C storage due to high concentrations of non-crystalline minerals associated with ash deposits [21−24]. Thus, this location provides an ideal system for assessing the inclusion of soil C tradeoffs within a renewable fuels production system, including the direct assessment of change in soil C stock due to conversion of pasture to managed eucalyptus forest.

Understanding the underlying processes driving soil C accumulation and change during land conversion is challenging [25], but should be integral to decision-making processes behind the implementation of sustainable, renewable fuel production systems [14,9]. Integration of C resources into system projections is particularly important where a C market is active and a pecuniary value can be assigned in addition to the CO_2 equivalence credit [17]. In this work, we used a recently established eucalyptus forest on previous pastureland in Hawaii as a case study to assess the C implications of pasture converted to eucalyptus versus pasture remaining in pasture. We focused on three sets of inter-related scientific questions:

- What is the current C balance of the managed eucalyptus forest and how does the C stock within aboveground tree biomass compare to the belowground biomass (roots) and soil?
- Did the soil C stock change following the conversion of pasture to managed forest? What were the primary ecosystem drivers of the change in soil C stock, if one occurred?
- What is the net climate change mitigation potential for eucalyptus forests along the Hamakua Coast on the island of Hawaii?

To answer these questions, aboveground biomass C in eucalyptus stands was measured, root biomass C was determined using allometric equations, and soil carbon stocks were measured to a depth of 1 m. We used two common methods of quantifying soil C stock in the top 1 m and compared their utility to assess land use change. To assess the implications of land use change, we measured C stocks under a series of paired pasture and eucalyptus plots. Finally, we compared the forest-level carbon values to the non-renewable GHG emissions for the production system. Our analysis provides a complete assessment of the C implications for the conversion of pasture to managed forest for a Hawaii case study. Within this case study, we highlight several practical challenges remaining to the fair and accurate implementation of C credits and trading schemes.

Methods

Soil and site description

The study area is located on the Hamakua coast in the northern part of the island of Hawaii, along an elevation gradient of approximately 600 to 1300 m (Figure 1). The study area receives mean annual rainfall ranging from 695 to 2590 mm and the mean annual temperature ranges from approximately 15 to 19 °C [26]. As elevation increases, precipitation and temperature decrease throughout the site. Soils are volcanic ash-derived, overlying lava flows that occurred 64,000 to 300,000 years ago, and are classified either as Hydrudands or Hapludands (Table 1). A managed *Eucalyptus grandis* forest (referred to hereafter as "eucalyptus forest") of approximately 2600 ha (19°58′N, 155°30′ W to 20°02′N, 155°27′W) was converted from kikuyu and pangola grass pasture (referred to hereafter as "pasture") and planted with *E. grandis* between 2001 and 2005. The afforested area was ripped to a depth of 18 to 24 inches (approximately 45 to 60 cm) and mounded with a

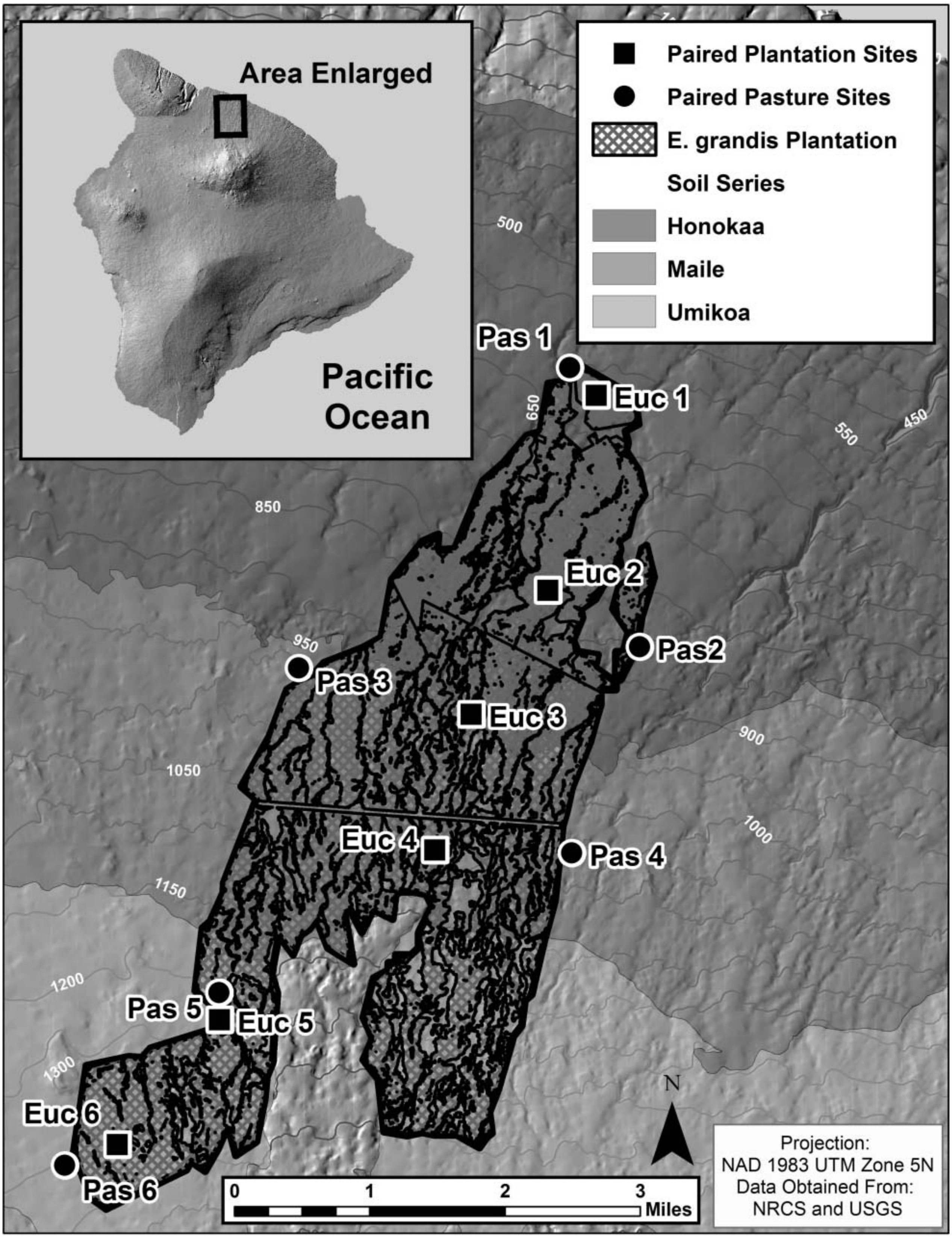

Figure 1. **Field site location on the windward coast of the island of Hawaii (inset) showing the current managed eucalyptus forest boundary overlaying soil taxonomic layers.**

disk plow to prevent air pockets in the seedling root zone. Six permanent sample plots (PSPs) in the eucalyptus forest have been continually monitored by Forest Solutions, Inc., since establishment. Measurements such as diameter at breast height (DBH, 1.4 m above soil) and total tree height have been taken annually for every tree in each of the 18 × 18 m PSPs. Resulting from this continual measurement, tree growth and yield for *E grandis* on this site is well understood and these PSPs serve as the basic

Table 1. **Elevation, mean annual temperature (MAT), mean annual precipitation (MAP), soil series and taxonomic description of soil series, and stand age for the six paired pasture (Pas) and *Eucalyptus grandis* (Euc) plots.**

Plot pair	Elevation (m)	MAT* (°C)	MAP** (mm)	Soil series	Taxonomic description	Stand age (years)
1	609/655	19.3	2591	Honokaa	Hydrous, ferrihydritic, isothermic Acrudoxic Hydrudands	7
2	877/823	18.0	2190	Honokaa	See above	7
3	888/986	17.6	2056	Maile	Hydrous, ferrihydritic, isothermic Acrudoxic Hydrudands	10
4	1102/1092	16.6	1294	Maile	See above	10
5	1239/1246	15.9	853	Umikoa	Medial, amorphic, isothermic Dystric Haplustands	10
6	1319/1333	15.3	696	Umikoa	See above	9

Note: *Rainfall values from Giambelluca et al. [26].
**MAT values from personal communication with T. Giambelluca.

experimental units for the eucalyptus forest plots in this study.

In 2011, six 18 × 18 m plots were established in pastureland adjacent to the eucalyptus forest plots at approximately equal elevation and in the same soil series (Table 1). The pasture plots were selected in areas that were relatively flat or gently sloping, to most closely match the eucalyptus forest plots. Higher elevation pastures are dominated by kikuyu grass (*Cenchrus clandestinus*), a common perennial C_4 grass that is naturalized in Hawaii [27], and pangola grass (*Digitaria eriantha*). Aboveground biomass of grazed kikuyu grass is approximately 2 Mg ha^{-1} [28]. At lower elevations, guinea grass (*Megathyrsus maximus*, previously *Urochloa maxima* and *Panicum maximum*), a productive and highly invasive species that is increasingly common in Hawaii, dominates and was present only in the lowermost site. Guinea grass live biomass ranges from 0.85 to 8.66 Mg ha^{-1} on Oahu [29]. Cattle are currently grazing all pasture sites.

Estimates of forage production at the pasture sites were derived using the Hawaii Forage Production Estimator tool [101]. Monthly forage supply for each plot (Mg dry matter ha^{-1}) was calculated based on monthly average rainfall values [26] (Table 1). Production values were then used to estimate the aboveground and belowground biomass C and annual root input. First, annual biomass production was multiplied by 0.50 to convert biomass to C. When these pastures undergo grazing, the cattle generally remove approximately 50% of the standing biomass [Thorne M, State Range Extension Specialist in the Kamuela Extension Office on the Island of Hawaii, Pers. Comm.]; therefore, aboveground biomass C of grazed pastureland was estimated by taking 50% of the production values. Root mass C was calculated as 80% of C in aboveground biomass [30]. Grasses typically turn over 30% of their root mass annually [30], which may be compounded by grazing. Here, grazing was considered to be 50%; therefore, an additional 2% loss was added to the natural die-off estimate of 30%, then multiplied by the root mass to get root litter input [101].

To enable the comparison of soil C stock and dynamics between the soils of the paired eucalyptus forest and pasture plots, the following was assumed: past land use was the same before the establishment of eucalyptus forest (both the pasture and eucalyptus are located on land that was converted from native forest dominated by *Metrosideros polymorpha* and *Acacia koa* 50–80 years ago); the soil series is the same for each of the pairs; thus, parent material, mineralogy, regional topography and weathering are equivalent; the elevation, precipitation and temperature are comparable; and local topography is similar, as all plots are located on flat to gentle slopes.

Above- and belowground biomass and carbon

Aboveground (ABG) biomass and C were computed from the measurements of tree height and DBH taken in 2011. Equation (1) was used to calculate ABG biomass:

$$ABG_{total} = 0.069413 * DBH^{2.1472} * H^{0.3129} \quad (1)$$

where ABG_{total} is the total dry weight of aboveground biomass in kg; DBH is diameter at breast height (1.4 m) in cm; and H is height in meters [31]. This equation was developed in Hawaii from the destructive harvest of *E. grandis* on the islands of Hawaii and Kauai. To convert to C, the ABG was multiplied by 51.1%, the C content of the eucalyptus trunk [102]. Belowground C (i.e., root biomass C) was calculated as 20% of ABG biomass C [32–34]. No shrubs and little to no ground cover other than sparse invading grasses were present.

Soil C stock

Soil cores were taken from five points (the four corners and the center of a square) in each of the 18 × 18 m plots. This design was based on intensive sampling of one test plot, from which it was determined that the sample data were independent at distances ≥ 10 m [35]. Five 1-m soil cores were taken per plot in 18-cm depth increments using a slide hammer corer of 4.39 cm diameter. The soil samples were split in half vertically, weighed and stored in a freezer. Half of the core was used to compute C stocks and the other half was used for soil fractionation. Bulk density was calculated from the mass of the entire core. Soils were oven-dried at 105 °C, sub-sampled, homogenized, ground to pass through a 250-μm sieve and analyzed for C content by elemental analysis (Costech Inc., CHN Analyzer).

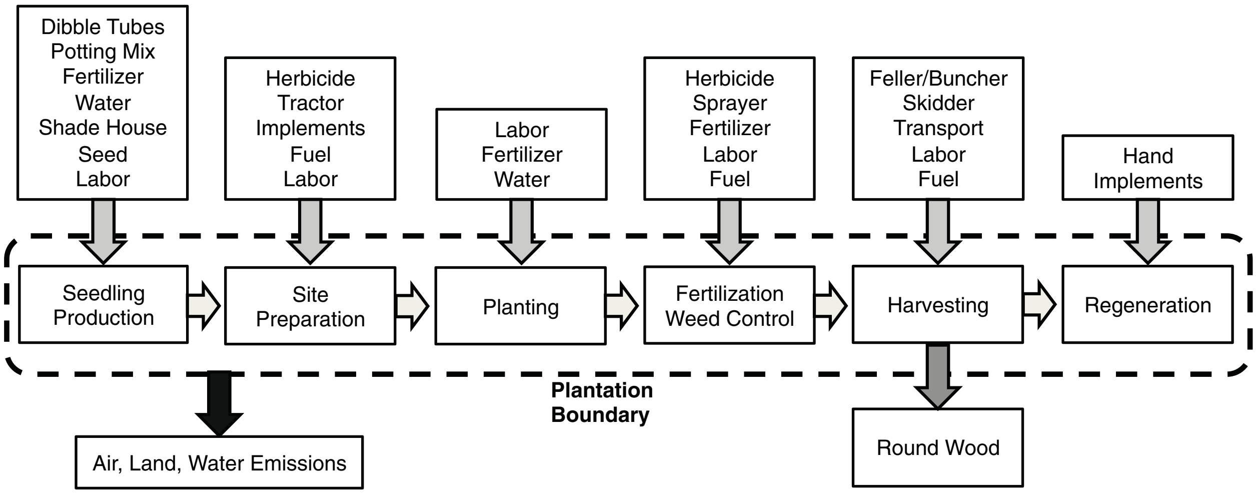

Figure 2. **Schematic of the life cycle inventory serving as the basis for calculating the global climate change impacts of anthropogenic activities, adapted from Turn et al. [38].**

Soil C stock (g cm^{-2}) was computed by two methods. The first method (see Equation 2) multiplied the C concentration by bulk density:

$$C\ stock = \sum_{i=1}^{n} (D_b * L_{core} * \%C/100)_i \qquad (2)$$

where D_b is the bulk density (g soil cm^{-3} on a dry weight basis); L_{core} is the length of the core (18 cm per section of the profile); %C is the concentration of C (g C 100 g $soil^{-1}$), and n is the number of soil cores. The C stocks calculated from the five sample cores in each plot were averaged and used to calculate one value for the amount of soil C per hectare for each site.

Although the method employing Equation (2) has been the convention for many years, this technique does not take into consideration compaction, expansion or erosion [36]. As a result, the fixed-depth method consistently overestimates organic C stocks [37]. A more rigorous method gaining recognition for quantifying C stocks is the equivalent soil mass (ESM) method. This methodology estimates C stocks using cumulative mass coordinates as opposed to spatial coordinates. By the theory presented in Wendt and Hauser [37], a fitted curve, in this case a quadratic polynomial, was used to calculate the organic C mass in a reference soil mass (see supplemental Figure 1).

Initially, the soil C stock was calculated by both methods to determine whether a difference existed between the two values for this system. Following the initial assessment, the remaining analyses and comparisons were conducted using the second, equivalent soil mass method.

Statistical analysis

Paired Student's *t*-tests were used to compare means of pasture versus eucalyptus plots for total C stock as well as C stock, bulk density and C concentration at each depth. The variance in C stock between the six pasture plots as well as between the six eucalyptus plots was calculated to compare variability within and between sites.

Life-cycle analysis

Turn et al. [38] performed a net energy analysis on the *E. grandis* production system practiced in Hawaii with a system boundary defined as the edge of the forest (Figure 2). Mass and energy flows crossing this boundary were inventoried over a 7-year production rotation and unit factors were applied to convert mass and energy flows to a common energy basis, MJ ha^{-1}. This life-cycle inventory served as the basis for calculating the global climate change impacts of anthropogenic activities. The analysis was performed using SimaPro software by PRé Consultants (Anon. 2008) [103] and utilized the US Life-Cycle Inventory [104], life-cycle analysis (LCA) Food Database [105] and EcoInvent Version 2 [106] databases. The Tool for the Reduction and Assessment of Chemical and other environmental Impacts 2.1 (TRACI 2.1) was used to calculate global climate change potential that includes the effects of other greenhouse gases (e.g. CH_4, N_2O, etc.). TRACI 2.1, developed by the United States Environmental Protection Agency (EPA), utilizes calculation methodology provided by the Intergovernmental Panel on Climate Change [39,40].

Results

The uppermost 1 m of mineral soil in the eucalyptus forest contained nearly 10 times the amount of C that was quantified within tree biomass, including both aboveground and belowground components (Table 2). The mean aboveground tree biomass C for the

Table 2. **Soil carbon stocks (Mg C ha^{-1}) calculated by both the bulk density method (to 1 m) and the equivalent soil mass method (to 4000 g soil). Mean values (± one standard error) of the five soil cores quantified for each plot are reported for the pasture and eucalyptus plot pairs. The change in C stock between pasture and eucalyptus (Δ) and the percent change (% Δ) were calculated for each plot pair. Biomass C (Mg C ha^{-1}), both aboveground (AG) and belowground (BG) and the sum (AG + BG) also are reported for the eucalyptus plots.**

	Bulk density method (Mg C ha^{-1} in 1 m)				Equivalent soil mass method (Mg C ha^{-1} in 4 kg)				Biomass C (Mg C ha^{-1})		
Plot	Pasture	Eucalyptus	Δ	% Δ	Pasture	Eucalyptus	Δ	% Δ	AG	BG	AG + BG
1	429.4 ± 16.7	543.4 ± 43.6	114.0	26.6	450.4 ± 10.0	477.0 ± 19.8	26.6	5.9	55.0	11.0	66.0
2	486.5 ± 12.7	641.7 ± 44.6	155.2	31.9	481.3 ± 6.7	505.1 ± 4.8	23.8	4.9	66.9	13.4	80.3
3	431.7 ± 21.4	535.1 ± 22.8	103.4	23.9	447.0 ± 12.9	518.9 ± 21.6	71.8	16.1	68.7	13.7	82.4
4	479.6 ± 25.8	652.0 ± 26.9	172.4	35.9	545.9 ± 44.7	568.0 ± 40.0	23.1	4.2	60.4	12.1	72.5
5	663.9 ± 20.7	661.8 ± 34.9	−2.1	−0.3	680.4 ± 26.1	637.3 ± 28.3	−43.1	−6.3	51.1	10.2	61.3
6	618.2 ± 68.3	537.0 ± 24.7	−81.1	−13.1	540.8 ± 40.18	492.33 ± 14.4	−48.5	−8.9	41.2	8.2	49.4
Mean	**518.2 ± 40.5**	**595.2 ± 25.5**	**77.0 ± 40.2**	**17.5 ± 8.0**	**524.2 ± 23.4**	**533.1 ± 21.5**	**8.9 ± 18.9**	**2.6 ± 3.7**	**57.2 ± 4.2**	**11.4 ± 0.8**	**68.7 ± 5.1**

eucalyptus forest was 57.2 ± 4.2 Mg C ha^{-1} and ranged from 41.2 to 68.7 Mg C ha^{-1} among the experimental plots, while the mean belowground root biomass was 11.4 ± 0.8 Mg C ha^{-1} and ranged from 8.2 to13.7 Mg C ha^{-1}. In comparison, live aboveground biomass and root biomass of the grasses in adjacent pasture plots was estimated to contain 0.73 to 2.80 Mg C ha^{-1} and 0.58 to 2.24 Mg C ha^{-1}, respectively (Table 3). Using the traditional bulk density method, estimates of the mean C stock for the top 1 m of mineral soil for the eucalyptus forest ranged from 535.1 ± 22.8 to 661.8 ± 34.9 Mg C ha^{-1}, averaging 595.2 ± 25.5 Mg C ha^{-1}. Corresponding equivalent soil mass method estimates ranged from 477.0 ± 19.8 to 637.3 ± 28.3 Mg C ha^{-1} and averaged 533.1 ± 21.5 Mg C ha^{-1}.

Although the total soil C stock quantified in the top 1 m of mineral soil of eucalyptus forest was of similar magnitude for the traditional bulk density and the equivalent soil mass methods, the change in C stock between the pasture and eucalyptus land uses was strikingly different depending on the method (Table 2). Using the traditional bulk density method, the differences in C stock between the pasture and eucalyptus forest at the six elevations ranged from − 81.1 to 172.4 Mg C ha^{-1}, with an average of 77.0 ± 40.2 Mg C ha^{-1}. In contrast, for the equivalent soil mass method, the differences in C stock between the six paired pasture and eucalyptus forest plots ranged from − 48.5 to 71.8 Mg C ha^{-1}, with an average of 8.9 ± 18.9 Mg C ha^{-1}. These differences in C stock between the pasture and eucalyptus forest represented an average change of + 17.5% and + 2.6% during land use change for the traditional bulk density and equivalent soil mass methods, respectively.

Table 3. **Estimates of pasture productivity and input rates for C in biomass and root turnover.**

Plot	Annual biomass production (Mg ha^{-1})	C in aboveground biomass (Mg C ha^{-1})	C in root biomass (Mg C ha^{-1})	C input in root turnover (Mg C ha^{-1} yr^{-1})
1	11.20	2.80	2.24	0.72
2	9.38	2.34	1.87	0.60
3	8.40	2.10	1.68	0.54
4	7.80	1.95	1.56	0.50
5	3.72	0.93	0.74	0.24
6	2.92	0.73	0.58	0.19

Direct comparison of the alternative C stock methods by depth and mass increments revealed that the greatest discrepancy between methods occurred in the middle-upper portion of the soil profile (Figure 3). In the uppermost depth (0 to 18 cm) and cumulative mass (800 Mg ha^{-1}) sections the magnitude and direction of change in soil C due to changes in land use are similar for both methods – that is, for both C stock methods there is no difference between pasture and eucalyptus forest plots. However, in the 18 to 36 cm and 36 to 54 cm depth sections, the C stocks estimated using the traditional bulk density method were significantly greater in the eucalyptus plots compared to the pasture (p = 0.003 and 0.071, respectively; Figure 3a). These differences in C stock were driven by differences in bulk density between land uses and not by differences in their C concentration (Table 4): for the 18 to 36 cm and 36 to 54 cm depth sections, the bulk density was significantly greater in the eucalyptus forest plots than the pasture plots and the C concentrations were not different for any depth between the land uses. In contrast, using the equivalent mass method, only the greatest cumulative mass increment exhibited significant differences in C stock between the two land uses, with eucalyptus forest greater than pasture (p = 0.085; Figure 3b).

Variability in the C stocks determined by the equivalent soil mass method and in the change in C stock resulting from land conversion across the plots was high but not correlated to any factors associated with the elevation gradient (i.e., mean annual temperature or precipitation), biomass or stand age. These results are consistent with a study on ecosystem C allocation in an undisturbed, native forest located within 25 km of the present study [41,42]. In that study, soil C stocks in the 0–91.5 cm depth averaged 253 Mg C ha^{-1}, and ranged from 194–288 Mg C ha^{-1}. Carbon pools did not vary along an elevation gradient although net primary productivity, litterfall, belowground C flux and soil respiration differed as a result of mean annual temperature [43]. Instead, volcanic ash-derived soil mineralogy exerted strong stabilizing and homogenizing effects on the C sequestered in the soil profile along the gradient [22].

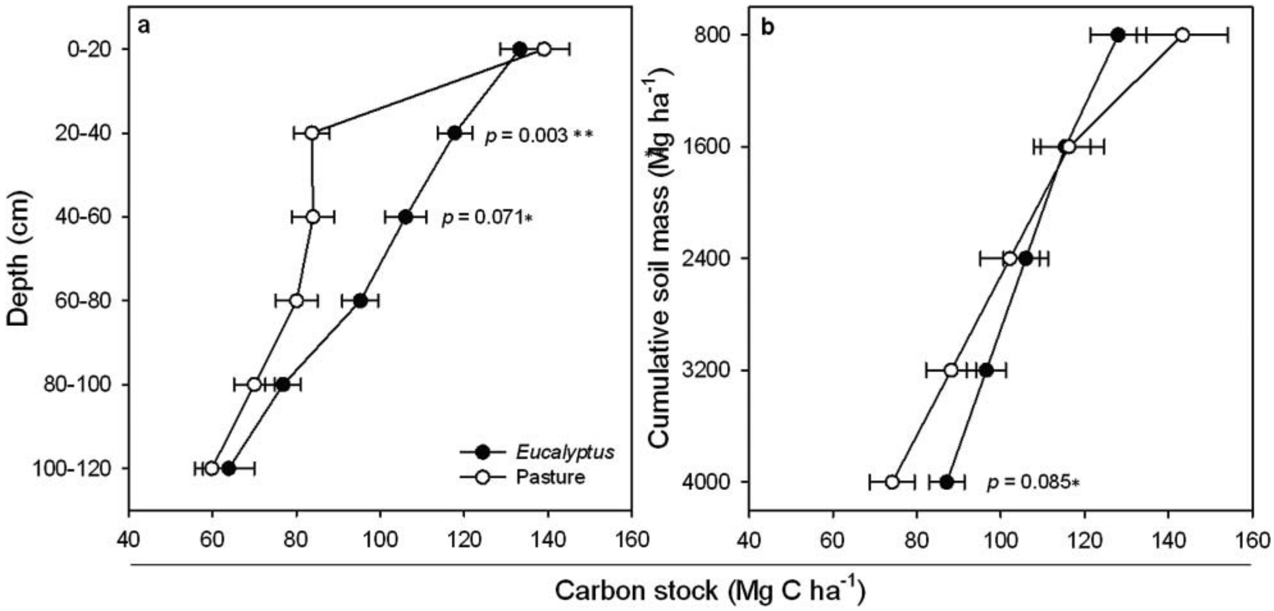

Figure 3. **Soil C stock (Mg C ha^{-1}) within each incremental soil depth (a) or mass (b); values are mean ± one standard error of the six plots for each pasture and eucalyptus pair. Significant Student's *t*-test results comparing the means between pairs for each depth or mass increment are shown; ** indicates $p < 0.05$ and * indicates $p < 0.10$; actual *p*-values are listed for significant differences.**

Using the forest boundary as the system boundary, analysis of the global climate change potential of anthropogenic activities supporting the eucalyptus production system showed that 3.7 Mg ha^{-1} of anthropogenic CO_2 eq is produced during one 7-year eucalyptus production cycle (Table 5). Diesel fuel used during harvesting accounted for 64.4% of the total global climate change potential, while herbicide and nitrogen fertilizer contributed 9.7 and 17.9%, respectively. With a simple conversion to account for the mass of C in CO_2, one 7-year eucalyptus production cycle generates 1.0 Mg C ha^{-1} in non-renewable emissions.

Discussion

In this case study system where the potential C storage is enhanced by soil mineralogy, the soil C resource in approximately the top 1 m, 533 Mg ha^{-1}, was nearly 10 times greater in magnitude than the C held within the tree biomass. This value is high, but consistent with other studies of similar soils [e.g. 41,42] and congruent with the very strong organic matter and clay mineral interaction characteristic to volcanic ash soils [21–24]. The measured value also is much greater than would be predicted by models such as the Rothamsted Carbon Model (RothC), which uses percentage of clay as the primary driver of soil C stock (although a recent module, RothPC, has been added specifically for volcanic ash-derived soils for this reason). Such models are commonly employed to predict soil C change in response to a land use or management change in some ongoing trials, for example the Kenya Agricultural Carbon Project that rewarded farmers for the adoption of SALM (Tennigkeit et al. 2013). Although the magnitude may not be as great in other locations, the soil always represents a potential resource for offsetting non-renewable aspects of bioenergy production [2].

Soil C stock, or the point-in-time quantification of the total amount of C within an area to a certain depth, is a fundamental metric for C resource accounting [44]. The method used to quantify soil C stock can affect the magnitude of the reported stock and the ability to accurately assess change in C stock as a result of land use change or management practice [36]. As reported

Table 4. **Soil carbon concentration (%) and bulk density (Mg m^{-3}) by depth for pasture and eucalyptus forest plots. Values are mean ± one standard error, pair-wise comparisons were made and *p* values reported for significant differences in mean values between plot type.**

Depth (cm)	% C			Bulk density		
	Pasture	Eucalyptus	*p* value	Pasture	Eucalyptus	*p* value
0–18	17.76 ± 1.44	16.16 ± 0.93	ns	0.46 ± 0.03	0.49 ± 0.03	ns
18–36	15.05 ± 1.10	14.53 ± 0.70	ns	0.35 ± 0.02	0.49 ± 0.01	< 0.001
36–54	13.59 ± 1.13	12.84 ± 0.69	ns	0.37 ± 0.03	0.48 ± 0.03	0.023
54–72	11.85 ± 1.17	12.23 ± 0.74	ns	0.40 ± 0.03	0.47 ± 0.03	ns
72–90	10.62 ± 0.99	10.61 ± 0.62	ns	0.39 ± 0.03	0.44 ± 0.02	ns
90–100	9.46 ± 0.85	9.77 ± 0.45	ns	0.41 ± 0.03	0.42 ± 0.02	ns

Table 5. **Global climate change potential of anthropogenic activities supporting *Eucalyptus grandis* production on a 7-year rotation using a forest-edge system boundary based on life-cycle inventory from Turn et al. [38].**

TRACI 2.1 impact category	System input	Unit	Global climate change potential (kg CO_2 eq ha^{-1} $rotation^{-1}$)
Seedling production			
Polypropylene, injection molded	0.58	kg/ha	2.6
Fertilizer (N)	0.24	kg/ha	2.2
Fertilizer (P_2O_5)	0.26	kg/ha	0.3
Fertilizer (K_2O)	0.28	kg/ha	0.2
Lime	0.11	kg/ha	0.0
Perlite, at mine	2.50	kg/ha	0.0
Peat, at mine	7.44	kg/ha	0.1
Transport, truck, average fuel mix/US	10.30	tkm/ha	1.0
Transport, ocean freighter, average fuel mix/US	49.97	tkm/ha	0.9
Transport, barge, average fuel mix/US	4.10	tkm/ha	0.1
Electricity, residual fuel oil	0.68	kWh/ha	0.6
Electricity, hydropower	0.05	kWh/ha	0.0
Electricity, geothermal	0.14	kWh/ha	0.0
Electricity, at wind power plant	0.11	kWh/ha	0.0
Site preparation			
Diesel, combusted in industrial equipment/US	14.03	l/ha	44.3
Transport, truck, average fuel mix/US	6.01	tkm/ha	1.0
Transport, ocean freighter, average fuel mix/US	7.12	tkm/ha	0.1
Transport, barge, average fuel mix/US	4.86	tkm/ha	0.2
Fertilization			
Fertilizer (N)	72.4	kg/ha	664.8
Fertilizer (P_2O_5)	28.8	kg/ha	34.0
Fertilizer (K_2O)	63.3	kg/ha	42.2
Transport, truck, average fuel mix/US	10.4	tkm/ha	2.8
Transport, ocean freighter, average fuel mix/US	2.6	tkm/ha	25.8
Transport, barge, average fuel mix/US	113	tkm/ha	3.7
Weed control			
Herbicides, at regional storehouse/RER U	35.1	kg/ha	358.7
Diesel, combusted in industrial equipment/l/US	33.7	l/ha	106.3
Transport, truck, average fuel mix/US	112.3	tkm/ha	10.4
Transport, ocean freighter, average fuel mix/US	144.4	tkm/ha	2.6
Transport, barge, average fuel mix/US	12.2	tkm/ha	0.4
Harvesting			
Diesel, combusted in industrial equipment/US	758	L/ha	2391.2
Transport, truck, average fuel mix/US	63	tkm/ha	5.8
Transport, ocean freighter, average fuel mix/US	14	tkm/ha	0.3
Transport, barge, average fuel mix/US	231	tkm/ha	7.6
Eucalyptus plantation total			**3710.2**

here, a direct comparison of C stocks derived from the bulk density and equivalent soil mass methods revealed a similar order of magnitude but substantially different assessment of soil C change as a direct result of land use change. If measured by the bulk density method, the conversion of pasture to managed eucalyptus forest increased soil C stock by 17.5%. If measured by the equivalent mass method, the conversion of pasture to managed eucalyptus forest increased soil C stock by only 2.6% (and was not significantly different from zero change). Thus, the choice of method for quantifying C stock in this system fundamentally alters the nature of the impact assessed on soil C stock – from a substantial to an insignificant gain.

The equivalent soil mass method specifically addresses the issues of compaction during land conversion/harvest and change in soil profile as C accumulates [36,45]. Compaction at upper-middle depths in the profile observed in eucalyptus stands in this case study was the primary factor in the difference between the C stocks derived by different methods. In the 18 to 72 cm section of the profile, C stock by the bulk density method was greater in the eucalyptus stands than the pasture. This shift contributed heavily to the overall gain in soil C with land conversion measured by this method. However, when change in bulk density was removed from the quantification using the ESM method, the increase disappeared. If the land use results in compaction, the bulk density method of determining soil C stock will overestimate the C sequestration attributed to the land use change. Reliance on this incorrect methodology to monitor and or verify C credits creates inaccuracy within the C accounting system.

The effect of depth on the difference in soil C stock as determined by the two methods reveals an additional limitation of currently accepted methods that quantify C stocks only to a shallow depth (often to 30 cm). Not only is the total C stock underestimated by limiting accounting to 30 cm, instead of to 1 m or deeper, but the overestimation of C stock change by compaction during land conversion is concentrated in the upper depths resulting in misleading or incorrect conclusions. Understandably, sampling soils to a depth of 30 cm is physically easier than sampling to 1 m. Richter and Billings (2015) reported that less than 2% of land use change studies sampled to 1 m [46]. The current results, however, demonstrate that the cost of inaccuracy in C credit and accounting may be great.

Biodiversity and functional diversity are ecosystem services critical for resilience to disturbance and climate change. Climate mitigation must be compatible with biodiversity and maintenance of diversified landscapes, both of which are critical to meeting global conservation and diversity goals [47]. In this case study, monoculture forest plantation replaced low-diversity, invasive/introduced grass pasture, resulting in no apparent change in plant diversity. Changes to the balance of the ecosystem were not quantified. At the landscape level, mean soil C stock alone does not capture the complexity of the soil system. Although the mean change in soil C stock was approximately zero, there was a wide range in values. Four of the six plot pairs gained soil C as a result of land conversion from pasture to eucalyptus but two of the six lost C, suggesting that a uniform approach to land management of this area may be inappropriate. These results demonstrate the potential for a sustainably managed, diversified landscape to provide substantial global climate change mitigation within a renewable energy system; however, further analysis is required.

When the soil C resource is considered a valuable component of the greenhouse gas balance of a system, and potentially as a pecuniary contribution through cap and trade market systems, then production decisions should include the protection and enhancement of this resource. Three important aspects of soil C resource assessment during land use change or land conversion emerged from this case study: method of resource assessment (i.e., bulk density versus equivalent soil mass), depth of sampling, and limitations in our understanding of the capacity for soils to sequester C and the ability to model soil C change. As part of a renewable fuels production system management strategy, maximizing gains in ecosystem C stocks on the landscape can provide a valuable offset to other non-renewable emissions incurred during the feedstock production, transportation and subsequent fuel conversion stages. In this case study of a managed eucalyptus forest established on previous pasture, plant productivity (but not soil C sequestration) resulted in C gains to offset non-renewable emissions at the forest boundary at approximately a 50 to 1 ratio, leaving substantial offset remaining to apply downstream to C costs incurred during processing for wood, bioenergy or biofuel products. As part of a renewable energy system, these net offsets may be credited as a climate change mitigation service through avoided fossil fuel emissions, and contribute substantially to lowering the global climate change potential of the production system.

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