Agriculture, Ecosystems and Environment 294 (2020) 106879

Contents lists available at ScienceDirect

Agriculture, Ecosystems and Environment

journal homepage: www.elsevier.com/locate/agee

Soil carbon stocks in Indonesian (agro) forest transitions: Compaction conceals lower carbon concentrations in standard accounting

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ARTICLE INFO

Keywords:
Agroforestry
Soil carbon
Carbon concentration
Bulk density
Carbon accounting
Climate change mitigation
IPCC

ABSTRACT

Soil changes matter for the global carbon (C) balance although belowground response to land use change is slower and less obvious than that aboveground. Impacts of changes from natural forest to a range of intermediate tree-based land uses (‘agroforestry’) and non-tree agriculture remain contested. Standard C-stock accounting for a fixed sampling depth depends on changes in both C_{org} concentrations and bulk density, often with opposite effects. Confounding factors that, beyond current vegetation, influence C_{org} (soil texture, minerology, drainage, elevation and soil pH) may also influence bulk density. Because land use may not be random with respect to inherent soil properties, differences in soil C-stock between land uses can have multiple causes. We compiled and analysed data from six landscapes in Indonesia (volcanic and other mineral soils; Sumatra, Kalimantan; Java, Sulawesi) where chronosequences of forest, various agroforestry systems and open-field agriculture had been sampled. Our data analysis (617 samples within 0 − 30 cm depth; 8 land use types) showed that a pedotransfer function for effects on C_{org} of texture, elevation and soil pH reduced the relative standard error of means per land use type, reduced the range (Max–Min)/Avg and led to a more consistent pattern in apparent land use effects. Relative to natural forest reductions in C_{org} concentration in the 0 − 30 cm layer (corrected for confounding factors) averaged 8–20 % in degraded forest, complex agroforest, oil palm plantations and older forest plantation plots, and 25–30 % in simple agroforestry, monoculture tree crops and woodlots, or over 40 % in non-tree (mostly cropped) plots. However, calculated C-stock change was small due to an observed increase (up to 30 %) of bulk density relative to that of natural forest. This implies that up to 23 % additional C_{org} became included in the soil sampling, resulting in a non-negligible bias (underestimate) in estimated soil carbon loss based on internationally agreed C-stock accounting.

1. Introduction

Soil is the largest terrestrial pool in the global carbon (C) cycle (Scharlemann et al., 2014; Banwart et al., 2014). Maintaining and where feasible restoring soil carbon stocks contributes to sustainable development strategies and to meeting the global commitment of the Paris Agreement to limit global warming to 1.5 °C (UNFCC, 2015; Minasny et al., 2017; Soussana et al., 2017; Baveye et al., 2018; Minasny and McBratney, 2018; Schlesinger and Amundson, 2019). If all the worlds’ soils could increase their carbon stock by 4‰ (or 0.4 %) yr^{-1} this would make a substantial contribution to the global climate change mitigation goals, but expectations need to be managed regarding the extent to which this is achievable (De Vries, 2018; Poulton et al., 2018). Active policies to incentivize increased soil carbon storage require understanding of the drivers of soil carbon decline, as well as support for soil management that leads to an increase (van Noordwijk et al., 2014).

Existing spatial variation in soil C-stocks is a combination of inherent properties and land use (Paustian et al., 1997; Post and Kwon, 2000; Guo and Gifford, 2002). Conversion of forest soils to agricultural use leads to a loss of soil-C, as organic inputs are reduced and decomposition rates increase with temperature in more exposed soils (Crowther et al., 2016; Melillo et al., 2017). A meta-analysis of published data (excluding peat soils) estimated a 25 % loss of soil C-stocks

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https://doi.org/10.1016/j.agee.2020.106879
Received 28 August 2019; Received in revised form 16 February 2020; Accepted 17 February 2020

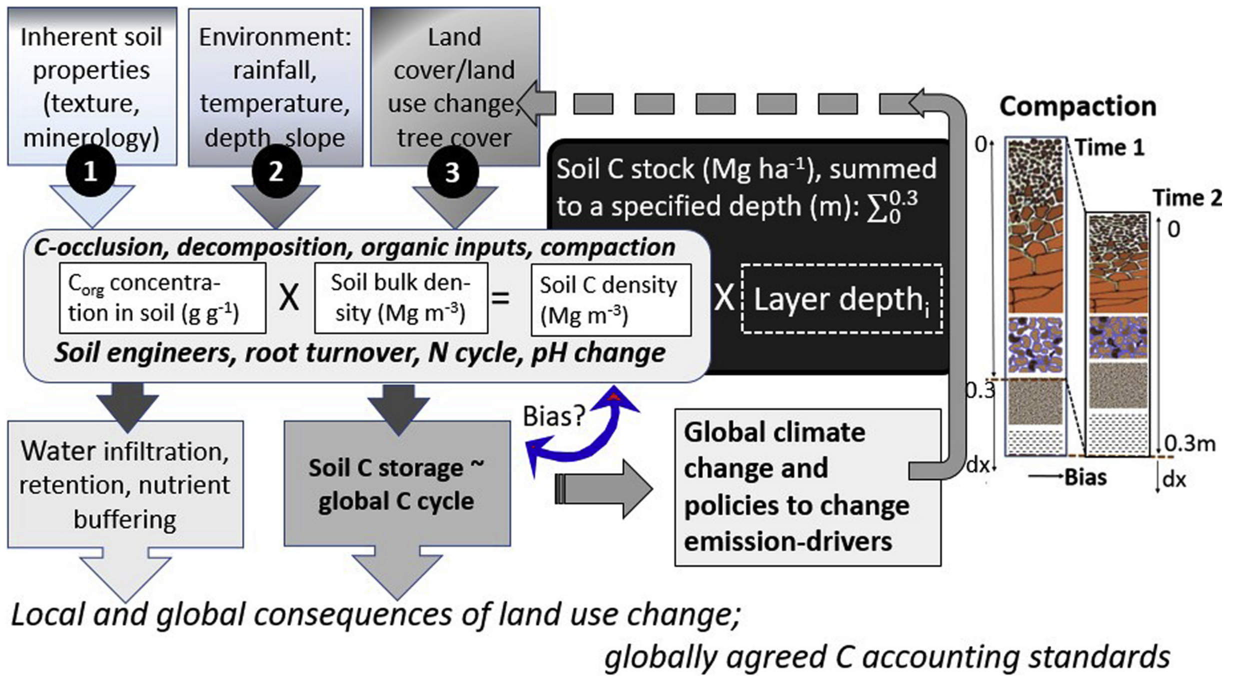

Fig. 1. Factors influencing soil C_{org} concentration, bulk density and their combined effect on soil C-stock, influencing local and global benefits and what is represented in a 0-30 cm depth sample of soil.

for the tropics (Don et al., 2011). Thus, "Agricultural soils, having been depleted of much of their native carbon stocks, have a significant CO_2 sink capacity" (Paustian et al., 1997). As restoring or increasing soil organic matter content is important for plant growth through effects on soil structure, infiltration, water retention and buffering of plant nutrient supply (Sanchez, 2019; Jackson et al., 2017), climate change mitigation and soil productivity may go hand in hand. A generic pattern of 'soil carbon transition' – shifts from decline to increase of soil carbon stocks without climate-related policy support has been documented (van Noordwijk et al., 2015) with similarities to the 'forest transition' concept for tree cover (Xu et al., 2007; Rudel et al., 2010; Dewi et al., 2017); if understood well, it can inform more targeted actions elsewhere. Increases in carbon stocks depend on both the amount of organic inputs (from above- and belowground biomass turnover; Rüegg et al., 2019) and decomposition rates (Fig. 1).

The IPCC guidelines for national greenhouse gas inventories provided a standard for quantifying and reporting belowground change in terrestrial C-stocks, focussed on the total amount of carbon in the top 30 cm of soil (Paustian et al., 1997; Eggleston et al., 2006). Soil C-stock varies with both C_{org} and bulk density (BD), with lowest BD values of around 0.05 Mg m^{-3} on some fibric peat soils associated with C_{org} values of around 50 % (w/w), and highest BD values of around 1.5 Mg m^{-3} on compacted mineral soils associated with C_{org} values of less than 0.5 %. C-stocks in the top 30 cm of soil peak in Andisols (BD around 0.6 Mg m^{-3}, C_{org} around 10 %) and mangrove Entisols with similar BD and C_{org}. Peatland C stock is special as it does not show the decline of C_{org} with depth found on mineral soils (except mangrove). When natural forests are affected by logging or converted to other land uses, with or without trees, the balance of organic inputs and decomposition shifts, resulting in a lower soil C_{org}. Within the IPCC accounting method, however, compaction of topsoil and a reduction of C per unit soil dry weight can counteract each other. After compaction a $0-30$ cm fixed depth soil sample includes soil particles (and associated soil C) that would not be included in samples prior to compaction e.g. they were below the sampling depth. Compression of soil can increase the carbon stock for a given sample, partly or fully offsetting a decrease in C_{org} (Fig. 1). Compaction can, through reduced infiltration induce surface runoff and erosion as secondary effect.

The aboveground biomass carbon stock increases after tree planting (and natural regeneration) (Requena-Suarez et al., 2019) with likely associated increases in root biomass and root turnover. However, the rate of input may not solely explain increases in C_{org} being also determined by clay and silt particles through physical-chemical processes that stabilize and protect carbon. More empirical evidence is required on the degree to which past losses of carbon and associated compaction can be undone. Recovery of forest-like soil hydraulic properties took 10–20 years after reforesting *Imperata* grasslands in the Philippines (Zhang et al., 2019). Recovery, beyond the topsoil in agricultural soils without trees may take even longer, as such soils lack a fresh source of old tree root channels (van Noordwijk et al., 1991).

Published estimates of the effect of tree crop monocultures on soil C-stocks vary widely (van Straaten et al., 2015, Khasanah et al., 2015). Agroforestry as land use practice has been reported to lead to soil properties intermediate between those of natural forests and intensively used agricultural lands, with details depending on the response parameter of interest and specific properties of the agroforestry system (including its rate of litterfall and root turnover (Saraiva et al., 2014; Rüegg et al., 2019), decomposition rates of above- and belowground litter, attractiveness for 'soil engineers' among the soil fauna). For specific agroforestry practices, such as fallows and multistrata agroforestry, and with soil that has been cropped for substantial periods of time there is evidence that the 4‰ yr^{-1} goal can be achieved (Corbeels et al., 2019). This may not be true for practices with lower tree densities such as alley cropping and parklands systems (Bayala et al., 2015). Reliable data on changes in soil C-stock in response to change in quality and quantity of tree cover–from natural forest, various forms of managed forestry and agroforestry (Fig. 2) – is needed for the IPCC national C accounting efforts, adjusting categories ('legends') to common land cover types and their functional differences in terms of C balance.

Forest soils under a permanent litter layer (Hairiah et al., 2006) are known for their loose structure, low BD and high C_{org} per unit dry weight of soil. Such soils facilitate water infiltration even during intense rainstorms, supporting root development and root functioning, as they rapidly regain aeration after short periods of water saturation. Soil compaction after conversion to agriculture, through the preferential loss of macropores, influences infiltration more than soil water

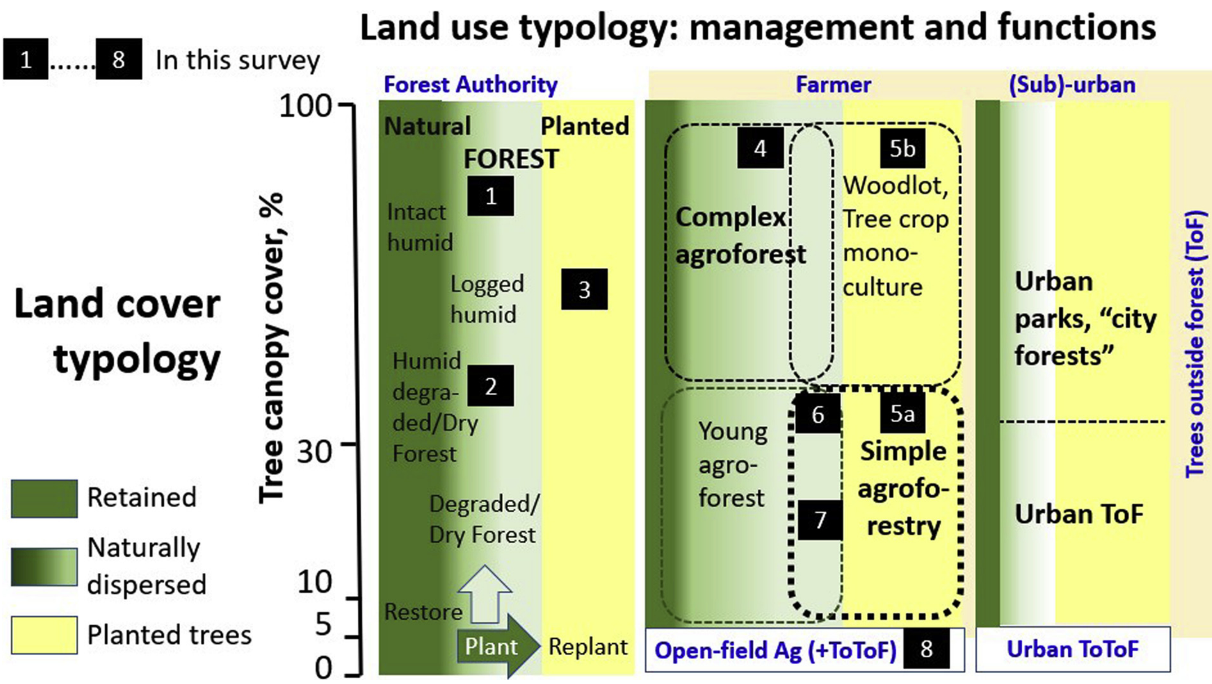

Fig. 2. Land use interpretation of land with varying degrees of tree cover based on the primacy of forest authorities (potentially delegated to communities or private enterprises), farmers (or agricultural plantations) or (sub)urban land managers; land uses included in the current chronosequences: 1. Natural forest; 2. Degraded forest; 3. Planted forest; 4. Complex agroforest; 5a. Oil palm; 5b. Tree monoculture; 6 and 7. Simple agroforestry and young agroforest; 8. Non-tree; ToF: Trees outside forest (with a minimum of 5% tree cover); ToToF: trees outside of trees outside forest, 0-5% tree cover.

retention (Wuest, 2009). Some agroforestry soils, however, maintain intact forest soil conditions by having a permanent litter layer, low degree of soil compaction and high organic matter content (van Noordwijk et al., 2019a). The dynamics of C_{org} and BD are partly responding to other properties of soil, climate and vegetation. The response to land use change to both C_{org} and BD needs to be teased apart from the background variation in soil type (with volcanic, peat and other wetland soils as special cases), soil texture (clay and silt contributing to occlusion of C_{org} in stable micro-aggregates, as well as influencing BD), pH, rainfall and elevation (as proxy for temperature) influencing decomposition and vegetation, and hence C_{org} rather than BD (van Noordwijk et al., 1997). As land use change is not random with respect to soil fertility (Sanchez, 2019), soil carbon stocks as observed cannot be interpreted as a one-way effect of land use practices. Rather, influences beyond land cover need to be accounted for as 'confounding factors' to get unbiased site-level estimates of the effects of land use change. Existing pedotransfer functions for C_{org} (van Noordwijk et al., 1997) and BD can be used for such.

The counteracting effects of change in C_{org} and BD has been noted before. Gifford and Roderick (2003) provided specific recommendations of on how to adjust sampling depth for changes in soil bulk density within a single study, but such adjustments require additional soil samples beyond what has been used in standard soil surveys for national IPCC C accounts. Various techniques have been used to correct for compaction in studies of land use change based on standardized sampling depths (Khasanah et al., 2015), but the issue remains to be addressed in IPCC guidelines for national C-stock inventories (Eggleston et al., 2006). For example, part of the measured increase in topsoil C-stock following 'conservation tillage' was due to an increased bulk density (and hence larger mineral soil mass) in soil samples to standard depth (Baker et al., 2007; Powlson et al., 2016). Similarly, part of the increase in C_{org} under organic farming systems with higher organic input levels is masked by a lower bulk density of the soil when measured at a standardized sampling depth (Gattinger et al., 2012). Lee et al. (2009) showed that when bulk density increases by surface soil compaction, it is safest to assume that originally deeper (and relatively low C_{org}) soil is included. McBratney and Minasny (2010) commented that the Lee et al. study did not use a proper mass coordinate system, in which soil C-stock is not affected by compaction, but which requires information about multiple layers in the same soil profile to relative cumulative soil mass to depth of sampling. Analysis of multiple land uses based on an 'equivalent soil mass per unit area' requires that there is a reference point for such soil mass – which the IPCC accounting method does not provide. In a global meta-analysis of 385 studies Don et al. (2011) concluded that without soil mass correction, land-use change effects on soil carbon stocks in the tropics would have been underestimated by 28 %. Most of the studies in that meta-analysis involved conversion from forest to agricultural use, the potentially subtler change between various forest and agroforestry types still deserve quantification and correction in default values as suggested by the IPCC accounting methods.

Indonesia with its rich diversity of soil types, globally high soil carbon stocks (Fig. 3) and wide range of agroforestry can provide insights of global relevance here. With 7.9 % of global agricultural land area, the Southeast Asia region represents 14.7 % and 28.9 % of global agricultural land with at least 10 % and 30 % tree cover, respectively (van Noordwijk et al., 2019b). Existing data suggest that a soil C transition is in progress, at least for the uplands of Java. Averaging over all Java-based samples of agricultural soils analysed at the Bogor soil research institute, Minasny et al. (2010, 2012) showed a consistent decline till around 1975, with highest loss rates in the 1950′s and 1960′s. After 1975 the rate of change became positive and part of the past losses could be recovered. In the period of 1930–1940 soil C in the $0-10$ cm depth layer was around 2 % (w/w); it declined to 0.8 % in 1960–1970 but increased again to 1.1 % around the year 2000. An increase from 0.8 to 1.1 % implies 35 years of an annual increase of 9‰ y^{-1} from the low baseline value (well above the 4‰ y^{-1} target discussed above). Soil C increases in this study were mainly related to changing agricultural practices: effective soil conservation and increased cropping intensity, increasing the root residue input per year. Sulaeman et al. (2013) summarized an Indonesian legacy database for mineral soils with agricultural use and found a mean C_{org} of 1.54 % and

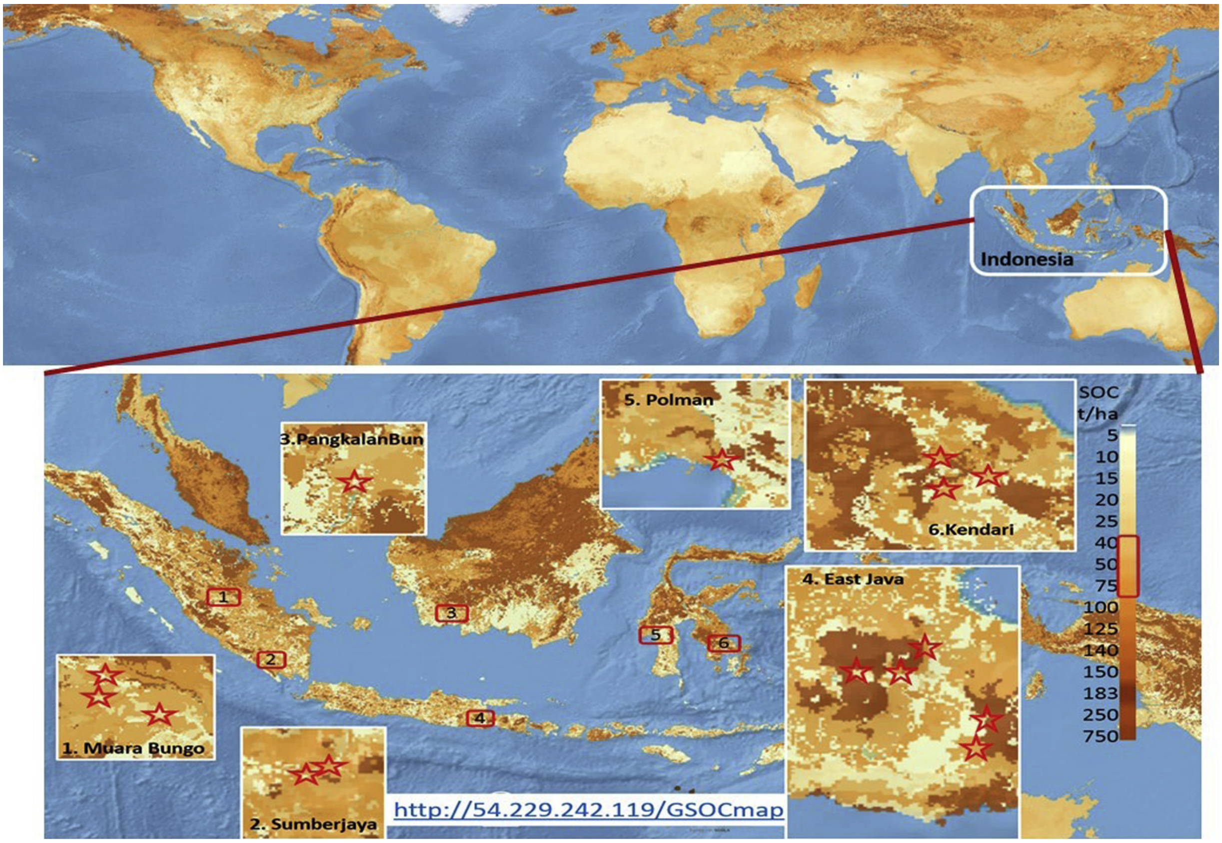

Fig. 3. Research locations in Indonesia superimposed on the global soil C-stock map with soil carbon in the 0-30 cm depth layer as available at the FAO GLOSIS - GSOCmap (v1.5.0) http://54.229.242.119/geoserver/GSOC/wms?.

mean bulk density of 1.34 Mg m^{-3} across the 509 and 474 samples. These estimates suggest an average C-stock for the top 30 cm of around 60 Mg C ha^{-1} but variation in both C_{org} and bulk density was substantial, with standard deviations of 1.45 % and 0.28 Mg m^{-3}. van Noordwijk et al. (1997) found no evidence for any change between soil surveys in the 1930′s and 1970′s in the non-linear relationship between soil pH and C_{org} for Indonesian forest soils, while soil carbon samples have been analysed with consistent methods since the 1930′s.

Making use of this diversity of soil types and the common presence of trees in agricultural land use ('agroforestry') in Indonesia we here synthesized a number of 'case studies' in various parts of Indonesia where coffee- and cocoa-based agroforestry had been compared to remaining forests and open-field agriculture. Some of these involved recent volcanic soils (classified as Andisols) with others having even more recent volcanic ash deposits (still classified as Entisol) or more mature Inceptisols. As categorical pedotransfer functions have not been tested in such a continuum of soil formation, we expected existing pedotransfer functions for soil carbon concentrations and bulk density to cover only part of the existing variation.

Specific questions for our data analysis were:

1 What is the mean relative change in soil carbon concentrations, bulk density and carbon stock in the 0 − 30 cm depth layer in Indonesian forest transitions from natural forest, plantations and agroforestry to agriculturally used soils across elevation ranges, volcanic and non-volcanic soils, and what is the confidence interval of these means?
2 To what degree do pedotransfer functions for reference C-org and bulk density reduce confounding factors in comparisons across land use types?
3 To what degree does, within the internationally agreed soil carbon accounting rules, soil compaction mask real changes?
4 How are changes in soil carbon related to aboveground biomass or litter?

The data used to address these questions have been compiled over the past twenty years. They compare 'chronosequence' land uses within a single landscape accounting for 'confounding' factors of non-random land use change to the degree that is possible.

2. Methods

2.1. Chronosequence sampling

Sampling followed the 'chronosequence' method in which spatial patterns of current land use systems are interpreted as indicative of temporal change from a common starting point (Sanchez, 2019. However, whilst similarity of initial conditions is controlled for as much as possible for site selection for such studies the explicit recognition of potential confounding factors is needed, through the use of pedotransfer functions as specified below. All data used for the current analysis were collected with the 'Rapid Carbon Stock Appraisal' method (Hairiah et al., 2011a, 2011b) or its predecessor (Hairiah et al., 2001). The method provides guidance on stratified sampling of local land cover classes in relation to understanding of land use systems and their life cycles (Fig. 2). It also provides protocols for sampling the five carbon pools specified by IPCC accounting methods, and data processing steps.

Within each landscape the locally dominant land cover types were sampled, including the best remaining forest and typical 'open-field' non-tree land cover types, as well as the various tree-based systems. Sampling site selection targetted land use types across a range available temporal variation. Soil samples for the various soil layers were composited from five sections of a 5 × 40 m^2 transect (Hairiah et al., 2011a, 2011b). In row crops with specified management zones such as oil palm

Table 1
General characteristics of the sampled landscapes and the number of samples per land use category included in the overall data set.

Landscape	Muara Bungo[1]	Way Besai watershed[2]	Arut watershed[3]	Malang and surroundings[4]	Polman[5]	Konaweha[6]
Province, Island	Jambi, Sumatra	Lampung, Sumatra	C.Kalimantan	E. Java	C. Sulawesi	SE Sulawesi
Soil types	Mineral, sedimentary soils: Ultisols, Inceptisols	Mostly Inceptisols, partly volcanic	Mineral, sedimentary soils: Ultisols, Oxisols	Partly volcanic landscapes: Inceptisols, Andepts, Andisols	Inceptisols	Inceptisols
Coordinates E	102°00′–102°22′	104°25′–104°27	111°44′–111° 51′	112°21′–112° 57′	119° 23′ -119° 24′	121°22′-122°31′
S.	1°00′–1°40′	5°01′–5°02′	2°24′-2°27′	7°05′-8°58′	3°25 - 3°26′	3°15′-5°13′
Elevation m a.s.l.	30–240	700-1700	20-50	1150; 1200; 560-1530; 290-1790; 430-2271	10-650	93 - 575
Mean annual precipitation, mm	2,000–3000	2550	2500	3150; 2005; 2584; 930–5500; 2360	2,113	1500-1900
Number of samples per land use:						
Natural forests	2	3		0, 0,0,24,0	0	0
Degraded forest	3	3	8	33,15,0,24,0,	9	9
Forest plantation	2	0		9,24,30,0,3	0	0
Complex agroforests	14	0		33,0,0,4,0	36	9
Simple agroforestry systems		5		33,0,30,4,8	0	9
Monoculture tree crops	2	3		9,0,9,4,24	12	9
Oil palm plantations	72	0	32	0,0,0,0,0	0	0
Open-field ('non-tree') crop systems	5	5		24,0,0,0,12	0	9
Total	100	19	40	141,39,69,60,47	57	45

[1] Muara Bungo (Jambi Province); Hairiah and van Noordwijk, 1997; Gillison et al., 2013; Saputra et al. 2019.
[2] Way Besai watershed, Sumberjaya (West Lampung); Dewi et al., 2006; Verbist et al., 2010.
[3] Arut watershed, Pangkalan Bun; Hairiah et al., 2011a).
[4] East Java: a) Kalikonto, (Ngantang); Hairiah et al., 2016, b) Kalisari (Karangploso-Malang) S slopes of Mt Arjuna; Mardiani, 2019, c) Welang (Prigen) E slopes of Mt Arjuna; Sari, 2010, d) Rejoso watershed (Pasuruan), N and W slopes of Mt Bromo and Mt Semeru; Hairiah et al., 2017, 2016, e) Bangsri sub-watershed, (Wajak-Malang), SW slopes of Mt Semeru; Kurniawan (2018).
[5] Binuang sub-district, Polman; Gusli et al. (2020).
[6] Konaweha watershed, Kendari; Saputra et al. (2019).

(Khasanah et al., 2015) samples were analysed per zone and used for a weighted average result at field level. Sampling depths varied between the various studies with most following the 0–5, 5–15, 15 – 30 cm depth protocol (Hairiah et al., 2011a, 2011b) with others using a 0–10, 10 – 30 cm scheme. Samples within a single chronosequence study were taken within a few weeks of each other, generally avoiding dry season conditions. The six landscapes in which one or more chronosequence studies were studied between 1997 and 2018 were all south of the equator (Fig. 3; Table 1). Locations are overlayed on the GlobalSoilMap (Arrouays et al., 2014) in Fig. 3.

Not all land use classes were present at all locations and as a result the mean elevation at which the land uses were sampled varied (Table 2), with natural forest found at the highest elevation and oil palm closest to sea level. The oil palm plots had the highest average clay but lowest silt content. All soils had a pH (1 N KCl) between 4.0 and 5.0 and were classified as acid.

Table 2
Soil and site characteristics for the eight land cover classes, across all landscapes.

Land use	Number	Elevation (m a.s.l.)	Sample midpoint (cm)	pH (1 N KCl)	Clay (%)	Silt (%)
Forest (nat)	29	1921	13.19	4.86	15.3	45.9
Forest (degr)	104	1088	13.53	4.56	16.2	43.2
Forest (plant)	68	949	12.28	5.00	14.6	43.9
AF-complex	84	446	14.23	4.24	17.0	33.8
AF-simple	101	653	13.12	4.60	14.2	35.3
Tree-mono	72	659	13.27	4.45	17.7	30.5
Oil palm	104	72	13.5	3.95	28.4	19.8
Non-tree	55	698	11.32	4.51	20.1	36.6

2.2. Land cover classes

For the current analysis data were grouped on the basis of vegetation (tree cover, spontaneous *versus* planted trees) as well as land use management (State, Farmers, Private sector) (Fig. 2). A visual impression for each of the classes (Fig. 4) relates to the mean aboveground biomass C and necromass (litter layer + dead wood where present).

The first three classes match the FAO forest definition: "… a land area of more than 0.5 ha, with a tree canopy cover of more than 10 %, which is not primarily under agricultural or other specific non-forest land use."

Natural forests are forests in which the natural regeneration cycle leads to a mosaic of stands of various age and successional stage, contributing to diversity at multiple scales.

Degraded forests are natural forests that have considerable damage, e.g. due to a high intensity of logging and/or exposure to fire. Regeneration based on natural processes can lead to secondary forests, that still differ in structure, function, species composition or productivity from natural forests.

Plantation forestry in production forest land, dominated by planted trees. In our samples this refers to mahogany (*Swietenia mahogani*) and pine (*Pinus merkusii*) plantations on Java.

The next four land uses are managed by farmers or agricultural plantation companies, and as such don't qualify as forests, even though tree cover meets the definition.

Complex agroforests combining planted and retained and/or naturally regenerated trees with a basal area of the main tree crop (e.g. coffee, cocoa or rubber) of less than 80 % and at least 5 tree species in a standard observation plot. In our surveys on Sumatra it includes jungle rubber where 'Para' rubber (*Hevea brasiliensis)* is planted (and/or spontaneously regenerating) in combination with local fruit trees (including duku (*Lansium domesticum*), durian (*Durio zibethinus*), rambutan (*Nephelium lappaceum*), jackfruit (*Artocarpus heterophyllus*), stink beans (*Parkia speciose*), jengkol (*Pithecellobium jiringa*), various timber tree

Fig. 4. Visual impressions of the land use classes and average aboveground C biomass + litter layer (Mg ha^{-1}) for the plots classified in each category.

species, and medicinal plants (Tata et al., 2008). Across the islands, it also includes coffee and cacao-based agroforest that meet the definition on relative importance and diversity of other trees; in these agroforests planted trees may dominate over spontaneously established and retained trees.

Simple agroforestry systems not meeting the basal area and/or species diversity criterion include coffee and cacao trees with a leguminous shade tree cover (*Gliricidia sepium, Leucaena leucocephala, Erythrina subumbrans*) and/or selected fruit trees (jackfruit, durian, avocado (*Persea americana*), duku) and timber trees. In some plots a ground cover of chili pepper (*Capsicum annuum*), taro (*Colacasia esculenta*), various vegetables or types of gingers were present e.g. *Zingiber officinale* (Ind: jahe), *Curcuma domestica* (kunyit), *Kaempferia galanga* (kencur), *Boesenbergia rotunda* (temu kunci), *Alpinia galangal* (lengkuas/laos), or *Elettaria cardamom* (kapulaga). Land use systems In East Java (Prigen sub district, Pasuruan district) with jackfruit and food crops (maize (*Zea mays*), taro) or salak (*Salacca zalacca*), banana (*Musa acuminata*) and papaya (*Carica papaya*) were also classified in this category.

Oil palm plantations consisting of monocultures of Elaeis guineensis.

Monoculture tree crops and woodlots in these surveys included coffee, cacao, and bamboo monocultures.

Finally, all non-tree systems were grouped as ***open-field ('non-tree') crops systems.*** This includes maize, cassava (*Manihot esculenta*), upland rice (*Oryza sativa*), soybean (*Glycine max*) and groundnuts (*Arachis hypogea*); at high elevations it includes vegetables, mostly carrots and cabbage. Land management is mostly intensive, involving soil tillage, planting, weeding, fertilizer (organic and inorganic) application, and pest control. Some plots were fallowed at the time of sampling, and some were covered by *Imperata cylindrica* after a cropping phase.

2.3. Temporal dimension

The chronosequences were sampled between 1997 and 2018 and are potentially influenced by three aspects of time: 1) changes in reference forest soil conditions over the twenty period, e.g. in response to changing forest management and atmospheric CO_2 concentrations, 2) different time periods since conversion of the various land uses at the time of sampling, 3) possible changes in the rate of soil C change in the land uses compared to forest, due to change in land use practices and atmospheric CO_2 concentrations. For the analysis we had to assume that aspects 1 and 3 are small relative to the changes we aimed to document and impacts of atmospheric CO_2 could be similar in direction and size between forests and comparator land uses. Aspect 2 is a potential source of bias as samples may not represent a balanced perspective on the whole life cycle of a forest-derived land use. For example, Khasanah et al. (2015) found evidence that the initial loss of forest soil C in newly planted oil palm is compensated by oil palm derived soil C if evaluated over the whole life cycle of the crop. We didn't have enough data to assess age effects within each land use type and accept that data may be biased towards relatively young (5 − 10 year old) versions of the land use systems.

2.4. Data collection and analysis

2.4.1. Overview of data processing

Before data could be analysed for possible effects of land use change two steps were needed to account for 1) confounding factors such as differences in elevation, sampling depth, pH, clay and silt concentrations, as clear from Tables 1 and 2) variation in depth intervals sampled within the 0 − 30 cm of soil. Both aspects were addressed by different parts of the overall 'pedotransfer' function. This aimed, for each sample, regardless of depth, texture, elevation or soil pH, to provide a reference value, C_{ref} and BD_{ref}, respectively that could be expected for a C_{org} or BD if the site had still been under prevailing natural forest. Analysis could then focus on the C_{org}/C_{ref} ratios. All measurements used the sample protocols and data processing of Hairiah et al. (2011a, 2011b), but some refinements had to be made to the pedotransfer functions used.

2.4.2. Pedotransfer function for C_{org}

The pedotransfer function for a reference soil C concentration, C_{ref}, expected for soils in Sumatra under natural forest conditions (as encountered in the soil surveys of the 1970's) was derived from van Noordwijk et al. (1997), and adjusted for overall level and modified for variable thickness of sample layers based on integration over an exponential distribution of C_{org} with depth:

$$C_{ref} = 0.9 \times (D_{Low}{}^{0.705} - D_{Up}{}^{0.705}) / (0.705 \times (D_{Low} - D_{Up})) \times EXP(A) \quad (1)$$

$$A = 1.333 + 0.00994 \times Clay\% + 0.00699 \times Silt\% - 0.156 \times pH + 0.000427 \times Elev + 0.834 \times Andisol? + 0.363 \times Wetland? \quad (2)$$

with D_{Up} and D_{Low} for the upper and lower depth of the sample (cm), pH standing for pH (KCl), Elev for elevation (m above-sea-level), and the Andisol? and Wetland? multipliers (0 or 1) for specified conditions.

The depth correction in the C_{ref} Eq. [1] was adjusted to the current data set, by minimizing a depth effect on C_{org}/C_{ref} across all land uses (note that these generally don't involve soil tillage). The underlying function (C_{org} = a $Z^{-0.295}$) was integrated from Z = D_{high} to Z = D_{low} to obtain the average C_{ref} concentration for a sample of any depth specifications. The depth term implied that soil C-stocks in the layers 0–5, 5–15, 15–30, 30–50, 50–100 and 100 – 200 cm depth could be expected to have 0.882, 0.516, 0.402. 0.338. 0.282, 0.230 times the C_{org} concentration (per unit soil dry weight) specified by the Exp(A) term in Eq. [1]. The power of the change of C_{ref} with depth, -0.295, is similar to that presented by Hartemink et al. (2010). It implies, in the absence of differences in soil bulk density, that 43 % of the soil C in the top 100 cm of the soil profile is found in the top 30 cm and that the 100 – 200 cm layer could add another 63 % to the total for 0 – 100 cm depth.

2.4.3. Pedotransfer function for bulk density

For bulk density three reference values, BD_{ref}, were calculated using two pedotransfer functions (the first in two variants). The first two made use of a texture-based pedotransfer for agricultural soils derived by Wösten et al. (2001) from a European data set. The first value, BD_{ref1}, estimated the bulk density at given C_{org} for a soil of the given texture:

$$BD_{ref1} = \text{IF}((\text{Clay\%} + \text{Silt\%}) < 50 \text{ THEN } 1/(-1.984 + 0.01841 \times 1.7 \times C_{org} + 0.032 \times \text{Topsoil?} + 0.00003576 \times (\text{Clay\%} + \text{Silt\%})^2 + 67.5 / \text{SandSize} + 0.424 \times \text{LN}(\text{SandSize})) \text{ ELSE } 1 / (0.603 + 0.003975 \times \text{Clay\%} + 0.00207 \times (1.7 \times C_{org})^2 + 0.01781 \times \text{LN}(1.7 \times C_{org}))) \quad (3)$$

with SandSize the median particle size of sand (default 290 μm), and Topsoil? A flag with 0 or 1 as values. For BD_{ref2} Eq. [3] was also used, but with C_{ref} substituted for C_{org}. It thus aimed to represent the bulk density of a forest soil of the same texture at its reference C_{org} concentration. A third reference for bulk density (BD_{ref3}) was derived within the current data set from a regression of BD on C_{org} as presented in Fig. 5 in the results section 3.2.

2.4.4. Estimating effects of land use change (Question 1)

2.4.4.1. Confidence intervals for changes in C concentration and bulk density. For data analysis C_{org}/C_{ref} and BD/BD_{ref} were calculated for each sample, before summarizing data for each land use class, across the various landscapes. In line with question 1, we focussed on means and their standard errors for the various land use classes (with 95 % confidence intervals of the means approximately + or – two times the SEM), rather than applying statistical tests of a null-hypothesis of 'no effects' with distributional assumptions that might not be easily ascertained for the data (given the use of ratios in the calculation of response variables).

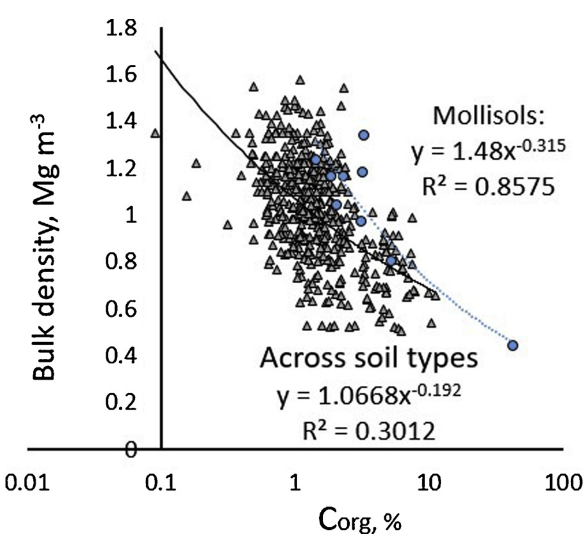

Fig. 5. Relationship between bulk density and C_{org} as basis for BD_{ref3} data across soil types; circle points and dotted regression line for Mollisols were derived from Shofiyati et al. (2010).

2.4.4.2. Confidence intervals for changes in C-stock within sampled 30 cm of soil. The 'observed' C-stock in the top 0 – 30 cm of soil was calculated as 30 × C_{org} × BD and compared to an 'expected' value as 30 × C_{ref} × BD_{ref}, for the three variants of BD_{ref}.

2.4.4.3. Correction factors for effective sampling depth. The ratio Observed/Expected is an indicator of C-loss but requires adjustment for the effective sampling depth of forest soil that would have provided the same amount of mineral soil as present in the sample. As a first step in this correction procedure, an equivalent forest sampling depth ($EqDepth_i$) was estimated for each land use *i* as:

$$EqDepth_i = 30 \times (BD_i / BD_{ref3,i}) / (BD_{NatFor} / BD_{ref3,NatFor}) \quad (4)$$

This equation implies that comparison is made on the basis of the constant soil dry weight expected for a forest soil under prevailing conditions.

The overestimate of the direct C-stock estimate due to higher soil bulk densities was then derived from Eq. 1 for a layer 30 to $EqDepth_i$ divided by the same for a sample layer 0–30 cm. Compensating for this bias, we obtained a corrected $Expected_{For,i}$ value.

Finally, Relative C-stock loss for the soil was calculated as:

$$\text{Relative C-stock loss} = 1 - Observed_i/Expected_{For,I} \quad (5)$$

With equivalent corrections for the Standard Error of the Mean.

2.4.5. Assessments for questions 2-4

Assessment of questions 2 and 3 was implied in changes in the relative confidence intervals of the mean in various steps of the use of pedotransfer equations. As confidence intervals of changes in confidence intervals are strongly dependent on assumptions of normality, we provide a qualitative description only of the observations as such.

For question 4, we compared regression of the relative C concentration (C_{org}/C_{ref}), relative bulk density (BD/BD_{ref3}) and C-$stock_{(0-30)}$ with average aboveground biomass across the eight land use classes with that on the average data litter plus necromass.

3. Results

3.1. C_{org} concentrations

Uncorrected for confounding factors direct measurements suggested that oil palm and natural forest had the lowest and highest C_{org} value, respectively, with the average of five 'non-forest' land uses being only 48 % of the value for the three 'forest' categories (Table 3). However, when the C_{ref} values are used as correction for confounding factors, the same comparison suggests an 83 % ratio. The C_{org}/C_{ref} ratio for AF-

Table 3
Means of C_{org}, pedotransfer reference value C_{ref} and the C_{org}/C_{ref} ratios, with standard errors of the mean (S.E.M.) for the eight land cover classes and the number of samples in each; range is calculated over the land use means as (Max - Min) / Avg.

Land use	Number	C_{org}, % *(S.E.M.)*	C_{org}/C_{ref}, *(S.E.M)*
Forest (nat)	29	3.25 *(0.393)*	1.000 *(0.109)*
Forest (degr)	104	2.39 *(0.200)*	0.925 *(0.049)*
Forest (plant)	68	2.71 *(0.232)*	0.810 *(0.039)*
AF-complex	84	1.51 *(0.117)*	0.881 *(0.047)*
AF-simple	101	1.26 *(0.051)*	0.746 *(0.039)*
Tree-mono	72	1.35 *(0.132)*	0.711 *(0.041)*
Oil palm	104	1.44 *(0.065)*	0.902 *(0.027)*
Non-tree	55	1.31 *(0.027)*	0.577 *(0.037)*
S.E.M./Mean		0.071	0.059
Range		1.118	0.515

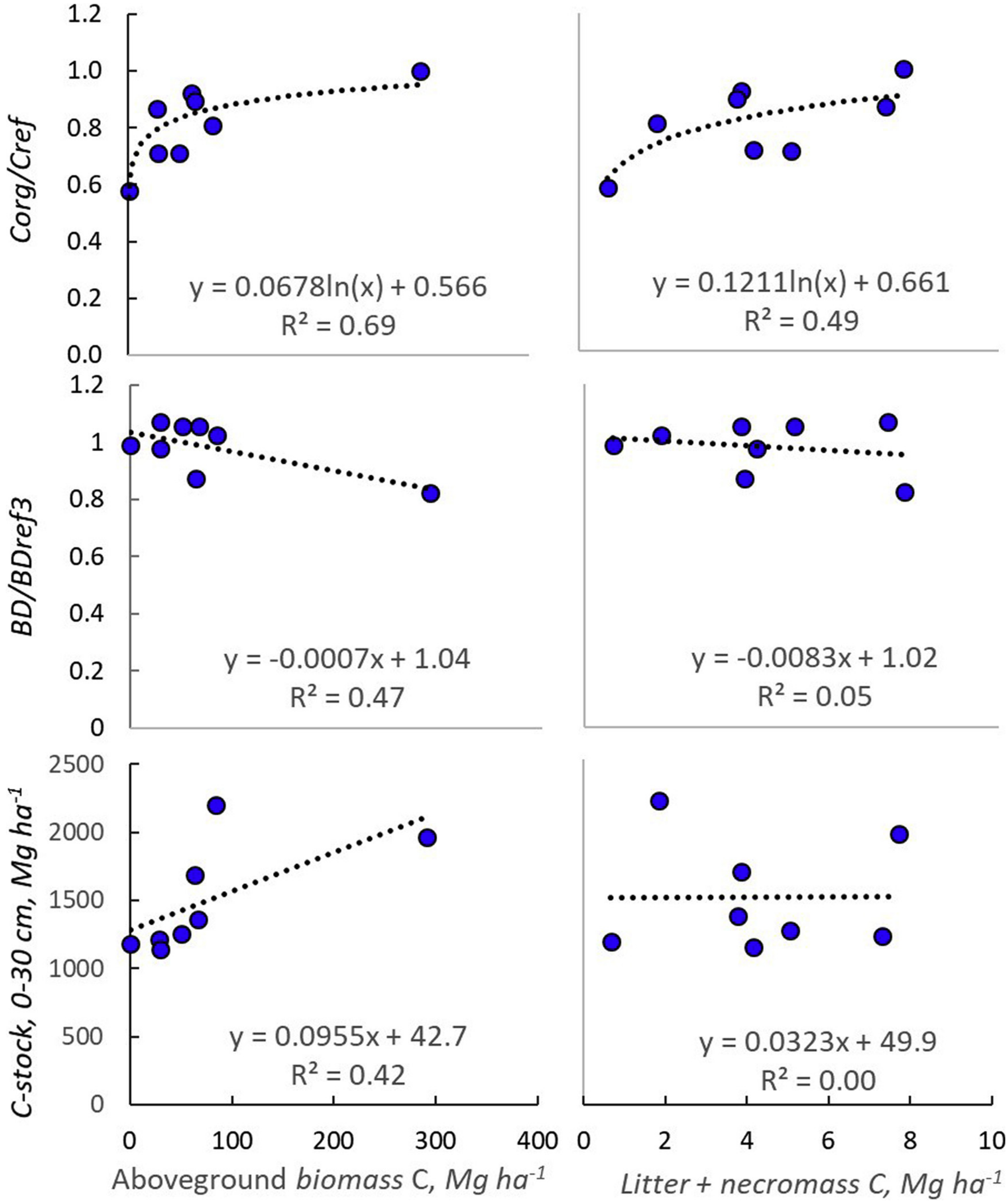

Fig. 6. Relationships between relative C concentration (C_{org}/C_{ref}), relative bulk density (BD/BD_{ref3}) and C-stock$_{(0\text{-}30)}$ with aboveground biomass (left-hand panels) and litter plus necromass (right-hand panels) across the eight land use classes.

complex is intermediate to plantation and degraded forests being 12 % below that of natural forest. Non-tree systems have a 42 % lower C_{org}/C_{ref} ratio than natural forests.

3.2. Bulk density

The lowest bulk density was recorded in natural forest and the highest in oil palm plantations. All values, however, were lower than the first two texture-based BD_{ref} values suggested for the given texture and soil organic matter contents. The BD/BD_{ref3} ratio, comparing bulk densities within the current data set, showed natural and degraded forest operated at a 18 % and 13 % lower bulk density, respectively, than the other land covers. Oil palm was still indicated as having the most compacted soils.

3.3. Aboveground bio- or necromass as predictor of soil C-stocks

Across the eight land use classes Fig. 6 compares aboveground biomass or litter layer (and dead wood necromass where present) as a predictor of the relative C_{org} concentration, relative bulk density and C-stock in the $0-30$ cm depth layer. Relations with biomass are stronger than those for necromass for all three properties. The relationship between biomass and C_{org}/C_{ref} is distinctly non-linear and a rapid increase in C_{org}/C_{ref} from land without trees to land uses with a biomass of $50-100$ Mg C ha^{-1} is followed by a slow further rise towards the values for undisturbed natural forest. The biomass equations would be similar for belowground biomass, as this is commonly estimated to be around 25 % of aboveground biomass. Some of the tree-based land uses with a biomass of $50-100$ Mg C ha^{-1} are indistinguishable from

Table 4
Bulk density results across land uses and three pedotransfer reference values: BD_{ref1} (based on soil texture and measured C_{org}), BD_{ref2} (similar, but based on C_{ref}), BD_{ref3} is based on C_{org} within this data set (Fig. 4); S.E.M. is the standard error of the mean; range is calculated over the land use means as (Max - Min)/Av.

Land use	BD, Mg m^{-3} (***S.E.M***)	BD/BD_{ref1} (***S.E.M***)	BD/BD_{ref2} (***S.E.M***)	BD/BD_{ref3} (***S.E.M***)
Forest (nat)	0.755 *(0.037)*	0.586 *(0.024)*	0.677 *(0.026)*	0.823 *(0.027)*
Forest (degr)	0.849 *(0.019)*	0.654 *(0.016)*	0.751 *(0.019)*	0.886 *(0.018)*
Forest (plant)	0.952 *(0.018)*	0.732 *(0.015)*	0.851 *(0.013)*	1.023 *(0.019)*
AF-complex	1.080 *(0.019)*	0.800 *(0.013)*	0.896 *(0.020)*	1.053 *(0.015)*
AF-simple	1.027 *(0.016)*	0.764 *(0.013)*	0.848 *(0.018)*	0.988 *(0.017)*
Tree-mono	1.099 *(0.021)*	0.818 *(0.015)*	0.922 *(0.024)*	1.053 *(0.018)*
Oil palm	1.125 *(0.024)*	0.858 *(0.017)*	0.898 *(0.019)*	1.105 *(0.023)*
Non-tree	1.038 *(0.037)*	0.779 *(0.020)*	0.901 *(0.025)*	0.988 *(0.027)*
S.E.M./Mean	0.0239	0.0227	0.0248	0.0211
Range	0.367	0.356	0.288	0.280

undisturbed natural forest in terms of soil C-stock (Table 4).

3.4. C-stock and bias in direct estimates for 0 – 30 cm layer

Using the measured C_{org} and BD values without corrections, C-stocks of 38–73 Mg C ha^{-1} were observed for the 0 – 30 cm depth layer. The AF-simple and Non-tree land use had a mean C-$stock_{0\text{-}30}$ below 40 t C ha^{-1} (Table 5). The three forest types had measured C-stocks of 56 – 73 t C ha^{-1}, but some of that may be due to their relatively high elevation. C-stocks expected for the given texture and elevation, using pedotransfer functions for both C_{org} and BD, ranged from 43 to 89 t C ha^{-1}. Relative to that expected value, oil palm had the highest C-stock.

Please note that the pedotransfer equation for C_{ref} was adjusted to match the mean of forest data in our data set, but individual forest datapoints maintain their variability, as reflected in the non-zero value for SEM for the forest category.

From the apparent increase in bulk density, an equivalent forest sampling depth can be calculated (highest, at 38.9 cm for oil palm, lowest 31.8 cm for degraded forest). This means that the initial (forest) C stock was underestimated by 4–20 % from the various land uses. After correcting for this bias, the last column of Table 5 presents our estimate of real soil C loss when current land uses are compared to the original natural forest condition. For non-tree land uses this loss is 24 % and for all agroforestry and derived forest systems the real C loss is between +10 % and -10 %, (+/- two times the estimated S.E.M.). These negative losses potentially indicate that our corrections for confounding factors are not yet fully effective and balanced. For AF-Simple, Plantation forestry and Tree-mono systems losses of 5–10 % are indicated (or potential gains over Non-tree of some 20 %) and for AF-comp a potential 10 % gain over AF-simple is indicated.

4. Discussion

Across the land use classes, we estimated that exclusion of soil bulk density changes underestimates changes in soil C-stock by up to 20 %. After the relevant corrections we estimated non-tree land uses to represent real soil C losses of 24 %, which is remarkably close to the result of a global meta-analysis by Don et al. (2011) of SOC losses by conversion of primary forest into cropland (−25 %). Where Don et al. (2011) found greater losses for conversion to perennial crops (−30 %), our data for Indonesia suggest that the prominence of agroforestry in tree crops such as coffee, cacao and rubber contributes to losses of only up to 10 %. Even smaller losses may apply to oil palm plantations when a life-cycle approach is used (Khasanah et al., 2015), consistent with the data presented here. Bulk density remains the weakest part in the chain leading to calculations of C-stock (Shofiyati et al., 2010), as current methods for measuring bulk density are laborious and expensive, subject to errors and complicated by the need to measure below the soil surface. Inclusion of bulk density data reduces the inferred changes in soil C-stock that are estimated from changes in C_{org} only.

Beyond global default values for soil C-stocks and the impact of land use change, higher tier methods have been defined by IPCC and must be well evaluated for a specified domain (e.g. climate region, soil type, crop type, topography), tailored to the land use transitions and management changes actually occurring in the area (Smith et al., 2012). As long-term site-specific repeated measurement data are scarce, 'chronosequences' (interpretation of co-occurring land cover/land use types as if they are the result of forest conversion) are accepted as basis for IPCC Tier 2 methods (Smith et al., 2012) but require measurements from multiple sites to avoid site-specific effects. Our data analysis is part of such effort, but not free from the confounding challenges of variation in the land use systems as classified under generic headings.

Our data documented a considerable spread in estimated soil C-stock in the top 30 cm, from 38 to 73 t C ha^{-1}. Such values match existing data for Indonesia (Fig. 3) and are relatively high, globally. Existing soil C maps reflect variation in soil and terrain properties, rather than current land use. Indeed, considerable efforts are needed to deal with confounding factors such as differences due to land cover (or land use), as our use of C_{ref} and BD_{ref} values demonstrated. A specific challenge to any interpretation of differences between forest, agroforestry and open-field agricultural soils in our data is the considerable difference in elevation, with intact forest absent or very scarce at low elevations, and protected forest most prominent on mountains in Java and increasingly in Sumatra as well. Elevation is a proxy for temperature in the tropics, with location-and scale-specific correlations between temperature and soil carbon sequestration quantified across the globe by Huang et al. (2018).

The specific form of the reference equations used adds to the uncertainty in our interpretation of results. While texture and pH effects

Table 5
C-stock estimates for the 0-30 cm depth layer in eight land cover types, the equivalent forest soil depth involved in the resulting 0-30 cm depth sample and estimates of real C loss through land cover transitions from natural forest; S.E.M. is the standard error of the mean; range is calculated over the land use means as (Max - Min)/Avg.

	Observed: $30 \times BD \times C_{org}$ (Mg ha^{-1})	Expected: $30 \times BD_{ref3} \times C_{ref}$ (Mg ha^{-1})	Observed/Expected	Equivalent forest depth (cm)	Initial C under-estimate (%)	Real C loss (%)
Forest (nat)	65.4 (*6.9*)	88.7 (*5.7*)	0.688 *(0.070)*	30.0	0.0	**0.0**
Forest (degr)	56.4 (*4.0*)	74.4 (*3.7*)	0.762 *(0.040)*	32.3	5.3	**−4.8**
Forest (plant)	73.3 (*5.9*)	84.4 (*4.9*)	0.768 *(0.040)*	37.3	16.5	**6.9**
AF-comp	45.4 (*2.2*)	51.7 (*1.9*)	0.913 *(0.040)*	38.4	19.0	**−7.4**
AF-simple	38.0 (*1.7*)	59.8 (*2.4*)	0.761 *(0.040)*	36.0	13.7	**4.7**
Tree-mono	42.0 (*1.7*)	59.6 (*2.4*)	0.764 *(0.040)*	38.4	18.9	**10.1**
Oil palm	47.4 (*2.1*)	46.9 (*1.1*)	1.006 *(0.036)*	40.2	23.0	**−12.5**
Non-tree	39.4 (*3.2*)	70.7 (*4.0*)	0.605 *(0.041)*	36.0	13.7	**24.2**
S.E.M./mean 0.065		*0.046*	0.059			**4.8**
Range	0.71	0.66	0.50			

documented in van Noordwijk et al. (1997) (the starting point of our C_{ref} equation) are consistent with global models of soil carbon dynamics, the volcanic nature of part of our soil, with partial andic features ('Andepts' for andic Inceptisols) but not classified as Andisols, was a major source of uncertainty. In this respect we were constrained by the underlying data that did not include a field test of andic properties. Generic pedotransfer functions for soil physical functions in relation to bulk density, C_{org} and texture can be improved on, once area-specific data are available (Rustanto et al., 2017), but global soil organic carbon assessments can still make considerable progress by re-using data that have been collected for other purposes (Stockmann et al., 2015).

Specific recommendations of Gifford and Roderick (2003) on how to adjust sampling depth for changes in soil bulk density require prior availability of site-level soil data rather than a standardized sampling as is commonly used. An approximation of the relationship between C_{org} and depth is unavoidable. Our samples referred to various soil depths but could through the adjusted C_{ref} equation be made comparable. A regression of the C_{org} /C_{ref} ratio on depth across all land uses accounted for zero percent of the variance (data not shown). However, extrapolating by use of generic multiplication factors for soil layers below 30 cm depth, will lead to further error and uncertainty. Geographic variation in soil C depth was linked to terrain properties in Gorontalo (Sulawesi, Indonesia) by Mason and Sulaeman (2016) but not land cover. They found that the spatial pattern of C-stock in the layer 30 – 60 cm depth was still proportional to that in the 0 – 30 cm layer, however, between 60 and 200 cm depth other sources of variation apparently become dominant.

Our approach assumes that compaction due to land use change takes place in the top 30 cm. On clay soils shrinking after deep drainage (as also occurs on peat soils) such assumption would not be valid. As far as we can judge, such conditions have not occurred within the data we presented here. Pedotransfer functions beyond texture have recently been discussed. Rasmussen et al. (2018) found stronger correlations of C_{org} with exchangeable Ca^{++} than with texture in water-limited, alkaline soils and with iron- and aluminium-oxyhydroxides in soils of higher moisture availability and acidity. However, causality is not clear in such a case, as exchangeable cations may in part be an effect of C_{org} and texture rather than a direct cause of C storage in soils. Exchangeable cation measurements were not part of our standard protocol so we cannot test for this. As step towards a more process-based understanding (and future modelling) of the variations in agroforestry and derived forest land uses shown in Fig. 6 are relevant. As fine root biomass is more closely related to leaves than to total aboveground biomass, the non-linearity of the relationship between biomass and C_{org}/C_{ref} can be expected whether above- or belowground inputs dominate. The weaker relationship between necromass and C_{org}/C_{ref} is consistent with the interpretation that roots rather than aboveground necromass and litter are the primary source of soil C (Saraiva et al., 2014; Rüegg et al., 2019). Estimates of design criteria for agroforestry by Young (1989) included a supposed need for around 8 Mg ha^{-1} y^{-1} of litterfall (with a 40 % humification rate) to compensate for the estimated annual decomposition rate of around 8 % y^{-1} of C-stocks of (Hairiah et al., 1992). Such rules need to be reconsidered incorporating belowground root turnover as additional and potentially more relevant source of soil C.

Our data and analysis were restricted to mineral soils, excluding the paddy rice fields, wetlands, peat areas (Khasanah and van Noordwijk, 2018) and mangrove (Atwood et al., 2017), all of which have specific forms of agroforestry and relatively high C-stocks. Further steps are required to inform emission factors for all forms of agroforestry in Indonesia. Although there still are expectations that evidence-based incentives at the farmer level could promote increased soil C-stocks, these increases are small considering the perceived transaction costs to make this a viable option (van Noordwijk, 2014). Agroforestry systems that maintain sufficient litter layers to protect the soil surface and have sufficient fine root turnover to maintain soil carbon are superior to open-field agriculture in soil C-stocks. The main current incentive for farmers to increase soil C content is formed by the increased buffering function for water and nutrients that such soils have, reducing exposure to climate variability and supporting adaptation.

5. Conclusions

Our four research questions thus lead to the following conclusions:

- The relative change in soil carbon stock in the 0 – 30 cm depth layer in Indonesian forest transitions from natural forest, plantations and agroforestry to agriculturally used soils amount to + or – 10 % for various tree-based land uses including agroforestry plantations and on average -24 % for non-tree land uses.
- Pedotransfer functions for reference C_{org} and bulk density reduce confounding factors in comparisons across land use types by ∼50 and ∼20 %, respectively.
- Soil carbon across land uses relates more closely to above- and belowground biomass than to standing litter stock, but the biomass found in agroforestry can bring soils close to forest soil conditions.
- Soil compaction (0–30 %) compensates for lower soil carbon concentrations (0–40 %), partially masking real changes in the internationally agreed accounting rules but causing a bias of up to 20 % in the IPCC methodology.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

This publication was prepared with support of WCU (World Class University) Program -SAMA (Scenarios Analysis for Management of Agroforests) activity 2019. Thanks to Drs. Ni'matul Khasanah, Nial McNamara and Rebecca Rowe for commenting on an earlier version of the manuscript. We also acknowledge the comments of two anonymous reviewers.

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