

## ECOLOGY

# Carbon neutral expansion of oil palm plantations in the Neotropics

Juan Carlos Quezada<sup>1,2\*</sup>, Andres Etter<sup>3</sup>, Jaboury Ghazoul<sup>4,5,6</sup>, Alexandre Buttler<sup>1,2,7</sup>, Thomas Guillaume<sup>1,2</sup>

Alternatives to ecologically devastating deforestation land use change trajectories are needed to reduce the carbon footprint of oil palm (OP) plantations in the tropics. Although various land use change options have been proposed, so far, there are no empirical data on their long-term ecosystem carbon pools effects. Our results demonstrate that pasture-to-OP conversion in savanna regions does not change ecosystem carbon storage, after 56 years in Colombia. Compared to rainforest conversion, this alternative land use change reduces net ecosystem carbon losses by 99.7 ± 9.6%. Soil organic carbon (SOC) decreased until 36 years after conversion, due to a fast decomposition of pasture-derived carbon, counterbalancing the carbon gains in OP biomass. The recovery of topsoil carbon content, suggests that SOC stocks might partly recover during a third plantation cycle. Hence, greater OP sustainability can be achieved if its expansion is oriented toward pasture land.

## INTRODUCTION

Rainforests have been a major source of land for newly established oil palm (OP) plantations, particularly in the main OP-producing countries such as Indonesia and Malaysia (1, 2). Conversion of rainforests to OP plantations negatively affects a number of ecosystem functions including ecosystem carbon (C) storage, soil fertility, and biodiversity (3, 4). In Southeast Asia, a global hot spot of greenhouse gas emission from deforestation and land use change, deforestation for OP cultivation was the second largest source of CO<sub>2</sub> emission [~0.3 giga tons (GT) of CO<sub>2</sub> year<sup>-1</sup>] (5). Replacement of forested areas by OP plantations reduces ecosystem C storage by up to 173 Mg C ha<sup>-1</sup>, owing mainly to the abrupt loss of biomass (6, 7). Such ecosystem C losses are exacerbated when OP plantations are established on tropical peatlands, as has occurred in Southeast Asia (8, 9).

In response to the detrimental environmental impacts associated with deforestation, various deforestation-free land use change trajectories have been proposed for a more sustainable OP expansion, including the use of marginal lands and conversion of savannas and pasture areas (10, 11). Of these, the use of pasture areas has great interest given the vast amount of land under pastoral systems and their low biodiversity and biomass C stocks (3, 12, 13). This land use change even has potential for climate change mitigation benefits by increasing C sequestration, given the substantially higher above-ground biomass C of OP over pastures (11). Yet, large uncertainties remain (2, 14), particularly with respect to changes in total ecosystem C (TEC) changes, soil organic carbon (SOC) stocks, and other soil

properties. Meta-analysis studies on the conversion of pastures into perennial plantations (often called afforestation) showed contrasting SOC stock changes (15, 16). Only two recent studies have quantified the effects of grasslands and pasture conversion into OP plantations on SOC storage, but their results are in disagreement (17, 18). Studies are much more numerous in the typical deforestation land use change trajectory of OP development on forested land. Most report that forest-to-OP conversion leads to soil degradation and C losses due to decreased organic matter (OM) inputs and erosion (4, 17, 19, 20), although a few have noted positive or unclear effects on SOC (17, 21, 22). In addition and despite the generally accepted view on SOC losses in OP plantations and the major role of soil organic matter (SOM) on soil productivity, soil chemical fertility and plant nutrient availability appear not to be negatively affected by OP agriculture, likely due to the use of mineral fertilizers in plantations (4, 23).

The reported changes in SOC following land use change to OP have occurred mainly in the surface layers (0 to 30 cm), and no effects in the subsoil have been detected. Yet, the focus of the literature to date has been the first OP rotation cycle (25 to 30 years) or even shorter time periods, which might not be a sufficient time to realize the effects of ongoing belowground processes. Evidence from other land use change types has already demonstrated that time after vegetation change (16, 24, 25) and sampling depth (26) have major roles on the magnitude and direction of changes in SOC stocks. The time following land use change needed for soils to reach either a new C equilibrium state or a subsequent recovery after the initial C losses usually occurs only after several decades (25, 26). In a recent comprehensive review on ecosystem functions in OP, the need to consider longer-term studies to assess the effects of OP cultivation on ecosystem functions was emphasized (4). The lack of observed responses in subsoil layers might be explained by the short time periods considered after land use conversion. Inputs of fresh OM from deep palm roots and nutrients from leaching might occur and stimulate the microbial mineralization of the large SOC stocks stored in subsoils (27, 28). The absence of change in SOC stocks in subsoils does not necessarily indicate that the old SOC stabilized during the previous land uses was not affected by the conversion, as it might instead be substituted by new SOC derived from OP inputs. This can be investigated when OP, which has a C3 photosynthetic pathway, is

<sup>1</sup>École Polytechnique Fédérale de Lausanne EPFL, School of Architecture, Civil and Environmental Engineering ENAC, Laboratory of Ecological Systems ECOS, 1015 Lausanne, Switzerland. <sup>2</sup>Swiss Federal Institute for Forest, Snow and Landscape Research WSL, Site Lausanne, 1015 Lausanne, Switzerland. <sup>3</sup>Department of Ecology and Territory, Pontificia Universidad Javeriana, Bogotá, Colombia. <sup>4</sup>Chair of Ecosystem Management, Institute of Terrestrial Ecosystems, Department of Environmental Systems Science, ETHZ, 8092 Zürich, Switzerland. <sup>5</sup>Prince Bernhard Chair for International Nature Conservation, Ecology and Biodiversity, Utrecht University, Padualaan 8, 3584 CH, Utrecht, Netherlands. <sup>6</sup>Centre for Sustainable Forests and Landscapes, University of Edinburgh, King's Buildings, Alexander Crum Brown Road, Edinburgh EH9 3FF, Scotland. <sup>7</sup>Laboratoire de Chrono-Environnement, UMR CNRS 6249, UFR des Sciences et Techniques, 16 route de Gray, Université de Franche-Comté, 25030 Besançon, France.

\*Corresponding author. Email: quezadarivera@gmail.com

established on tropical pastures or savannas, which are dominated by grasses having a C4 photosynthetic pathway (29). The different fractionation intensities of the two photosynthetic pathways result in differences in the isotopic signature ( $\delta^{13}\text{C}$ ) of the biomass, enabling the origin of SOC to be determined (30). This provides a powerful tool to study decomposition and stabilization of SOC after land use change.

In Colombia, the expansion of OP plantations has occurred mainly on pastures planted on cleared savannas and, to some extent, also on native tropical savannas. These land use changes contrast with the deforestation trajectories of the main OP-producing countries in Southeast Asia (14, 31). OP coverage in Colombia, which is currently the fourth major OP producer worldwide, has increased rapidly, tripling in the last two decades from roughly 160,000 to 480,000 ha (32). Of this expansion, about 60% involved the use of low productivity pasture areas (31). Nearly half of the land devoted to OP cultivation in Colombia occurs in the Llanos region of eastern Colombia, where cattle ranching is the main land use (33, 34), and future scenarios predict a fast growth rate of the OP industry in this region (31).

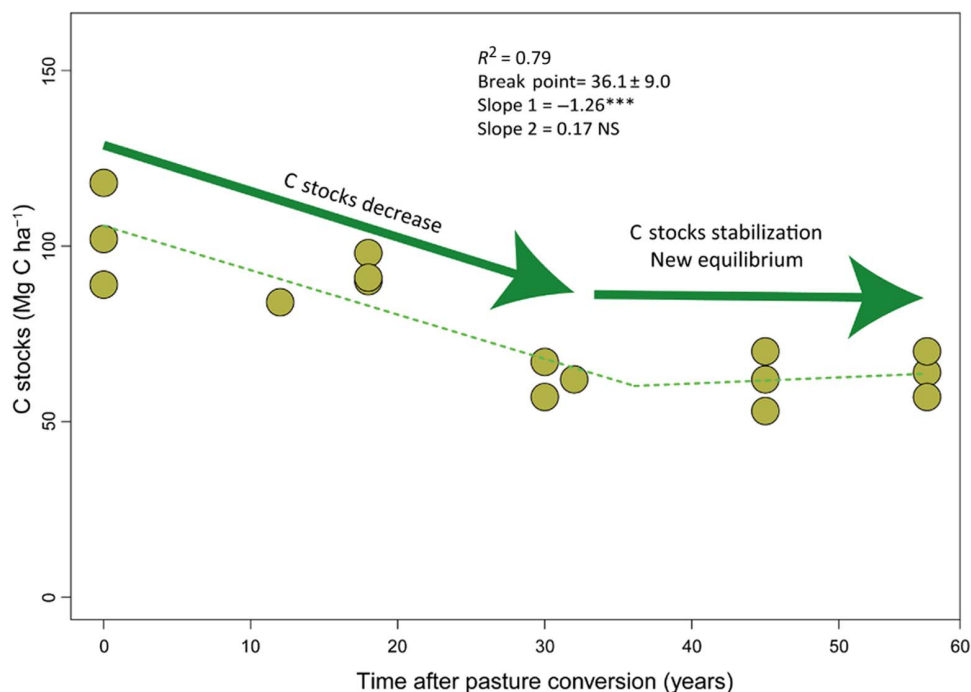
This study aims to assess the pasture-to-OP transition as an alternative to deforestation to mitigate the net TEC losses of future OP expansion. Specifically, we hypothesized that C gains in OP biomass might be counterbalanced by C losses in the soil. SOC stocks would be affected down to 50 cm, but a new equilibrium would be reached in all layers after two rotation cycles. Because of the long-term chemical fertilization in OP plantations, we expect to observe trade-offs among soil ecosystem services such as C storage and nutrient provision. To test these hypotheses, we quantified the dynamics of OP biomass C stocks, OP-, and pasture-derived SOC stocks down to 50 cm and nutrient availability over a long-term 56-year chronosequence of OP plantations established on former pastures in Colombia, taking advantage of a shift from C4 to C3 vegetation.

## RESULTS

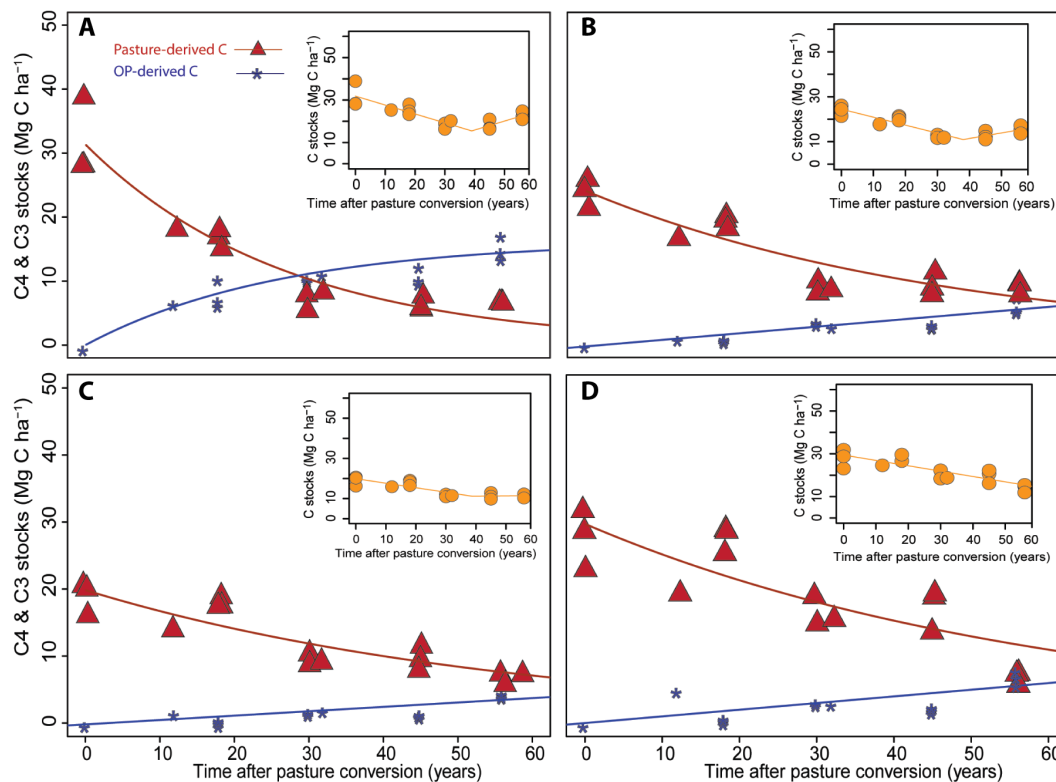
### Soil C stock dynamics

Cultivation of OP on former pasture areas severely affected SOC stocks to a depth of 50 cm (Fig. 1). Of the  $102 \pm 8 \text{ Mg C ha}^{-1}$  stored in pasture SOC  $39 \pm 8\%$  were lost after 56 years of OP cultivation. However, the C loss dynamics followed neither a linear nor an exponential trend but, rather, two trends in one adjusting to a segmented regression model (Fig. 1 and table S1). SOC stocks down to 50 cm constantly decreased until the beginning of the second OP cycle (break point,  $36.1 \pm 9.0$  years) at a rate of  $1.26 \pm 0.26 \text{ Mg C ha}^{-1} \text{ year}^{-1}$ , after which SOC stocks stabilized along the second OP rotation cycle.

The dynamics of the total SOC stocks down to 50 cm resulted from the combination of variable rates and patterns in the accumulation of OP-derived C and decomposition of pasture-derived C at different depths (Fig. 2). In the surface soil layer (0 to 10 cm), bulk SOC stocks decreased sharply at a rate of  $0.42 \pm 0.08 \text{ Mg C ha}^{-1} \text{ year}^{-1}$  until  $39.1 \pm 4.5$  years and then stabilized for the rest of the cultivation time. The initial decline in bulk SOC stocks was driven by a marked loss of pasture-derived C (77% after 39 years) that was not fully compensated by the accumulation of OP-derived C during the same period of time (Fig. 2A). The decomposition of pasture-derived SOC followed a first-order decay with a half-life time of 18.7 years ( $k = 0.037 \pm 0.0038$ ; Table 1). The accumulation of OP-derived SOC in the surface soil layer was best fitted by an exponential rise to equilibrium model, indicating a saturation in the accumulation of OP-derived C. The estimated gross OP-derived C input was of  $0.62 \pm 0.10 \text{ Mg C ha}^{-1} \text{ year}^{-1}$ , and its annual decay rate was of  $0.038 \pm 0.010 \text{ year}^{-1}$ . The decay rate constants of SOM between OP-derived and pasture-derived C were similar, suggesting no preferential C source for decomposers.



**Fig. 1. Soil carbon stocks after pasture conversion into OP plantations at 0- to 50cm depth.** The dashed line represents the fitted segmented regression equation. Significance of the slope coefficients from each side of the breaking point is indicated (\*\*\* $P < 0.001$ ).



**Fig. 2. Dynamics of pasture C4-derived C and OP C3-derived C following pasture conversion into OP plantations.** (A) 0- to 10-cm soil layer, (B) 10- to 20-cm soil layer, (C) 20- to 30-cm soil layer, and (D) 30- to 50-cm soil layer. Inset graphs show bulk SOC stocks for each layer. See Table 1 for the used functions and their kinetic parameters.

At break point (39 years), the net difference between the kinetic rates of change (according to the difference between the first derivative of Eqs. 4 and 5) was very low ( $-0.13 \text{ Mg C ha}^{-1} \text{ year}^{-1}$ ). SOC stocks in the top 10 cm would reach equilibrium at  $16.33 \text{ Mg C ha}^{-1}$  (calculated as division between  $A$  and  $k$  parameters of Eq. 4), according to the model based on C isotopes, i.e., similar level to the one estimated by the segmented regression based on bulk SOC ( $15.46 \text{ Mg C ha}^{-1}$ ; Table 1). These latter two estimations cross-validated the two approaches (bulk SOC and isotopic) used to estimate C dynamic in this work.

The SOC bulk stock dynamics in the two deeper soil layers (10- to 20-cm and 20- to 30-cm) exhibited similar patterns, i.e., a constant decrease followed by a stabilization of the stocks along the second rotation (Fig. 2, B and C, and Table 1). However, the C loss rates for these two soil layers were 14 and 46% less pronounced than those for the surface soil layer but stabilized at similar time:  $37.8 \pm 4.4$  and  $38.9 \pm 10.3$  years for the 10- to 20-cm and 20- to 30-cm soil depths, respectively. The decomposition of pasture-derived SOC also followed an exponential decay in the 10- to 20-cm and 20- to 30-cm layers. The decay rates, however, were lower, resulting in longer half-life of these pools, as compared to the surface layer (18.7, 34.7, and 40.8 years for the 0- to 10-cm, 10- to 20-cm, and 20- to 30-cm soil depths, respectively). In contrast to surface soil layer, the accumulation of OP-derived C did not show any saturation, with constant accumulation rates of  $0.10 \pm 0.01$  and  $0.07 \pm 0.01 \text{ Mg C ha}^{-1} \text{ year}^{-1}$  for the 10- to 20-cm and 20- to 30-cm layers, respectively, throughout the two rotation cycles (Fig. 2, B to D, and Table 1).

As indicated above, bulk SOC stocks reached an equilibrium in the 0- to 30-cm layer. In contrast, bulk SOC stocks in the deepest

soil layer (30 to 50 cm) were still decreasing at a rate of  $0.25 \text{ Mg C ha}^{-1} \text{ year}^{-1} \pm 0.04$  ( $R^2 = 0.71$ ; Table 1) after 56 years. This finding is of interest as no studies have reported, so far, effects of land use change to OP in subsoil horizons. The obtained C loss rates for bulk SOC stocks (slope 1 for three uppermost soil layers and linear regression slope for the deepest layer) decreased gradually with soil depth from  $0.42 \text{ Mg C ha}^{-1} \text{ year}^{-1}$  in the surface of 10 cm to  $0.13 \text{ Mg C ha}^{-1} \text{ year}^{-1}$  in the 30- to 40-cm and 40- to 50-cm soil depth (calculated from half of the depth of 30 to 50 cm).

### Ecosystem C stocks

OP plantations contain substantially greater total biomass (above- and belowground) than pastures. Total OP biomass increased at an accumulation rate of  $3.3 \pm 0.1 \text{ Mg C ha}^{-1} \text{ year}^{-1}$  along the 30 years of both, the first and the second OP cultivation cycles. This corresponded to time-averaged OP biomass C stocks of  $49.5 \pm 1.5 \text{ Mg C ha}^{-1}$ , i.e., five times more than  $10 \text{ Mg C ha}^{-1}$  typically found in the pastures of this region (35, 36). Although the SOC stock varied with plantation age, in general, soil was the largest C pool in the ecosystem. The contribution of SOC to TEC stocks ranged from 38% in the 30-year-old first cycle plantation to 87% in the recently replanted plantation 2-year-old second cycle plantation (32 years).

Time-averaged TEC in pastures reached  $112.8 \pm 8.3 \text{ Mg C ha}^{-1}$ , assuming constant SOC stocks ( $102.8 \pm 8.3 \text{ Mg C ha}^{-1}$ ) and a total pasture biomass of  $10 \text{ Mg C ha}^{-1}$  (Fig. 3). Over a large number of OP rotation cycles, time-averaged SOC stocks in OP plantations would be equal to the stocks at equilibrium ( $62.61 \pm 2.73 \text{ Mg C ha}^{-1}$ ; Fig. 1). Accordingly, time-averaged TEC in OP plantations

**Table 1. List of functions and kinetic parameters.** Parameters describe changes in bulk SOC stocks and C3- and C4-derived C stocks in the four sampled soil layers.  $R^2$ , coefficient of determination; slopes 1 and 2, significance of the two slopes for segmented regression analysis;  $K$ , rate constant;  $A$ , annual input of C3-C;  $C_0$ , SOC stocks before pasture change to OP. \*\* $P < 0.01$ ; \*\*\* $P < 0.001$ .

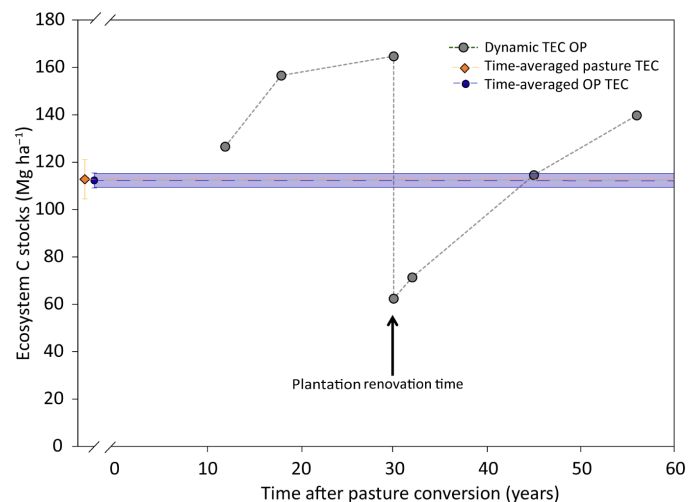
Soil layer	Model type	Function	$R^2$	Slope 1	Slope 2	AIC	$k$ (year <sup>-1</sup> )	Half-life	$A$	$C_0$
Bulk soil, 0- to 10-cm	Segmented	$F(t) = 31.87 - 0.42*t$	0.75	***	NS	88.26	—	—	—	—
OP-derived C, 0- to 10-cm	Exponential rise to equilibrium	$F(t) = -0.62*exp(-0.038*t) + 0.62/0.038$	0.91	—	—	66.0	**	18.1	***	—
Pasture-derived C, 0- to 10-cm	Single exponential decay	$F(t) = 31.4*exp(t* - 0.037)$	0.91	—	—	86.2	***	18.7	—	***
Bulk soil, 10- to 20-cm	Segmented	$F(t) = 24.49 - 0.36*t$	0.83	***	NS	75.52	—	—	—	—
OP-derived C, 10- to 20-cm	Linear	$F(t) = -0.27 + 0.10*t$	0.85	—	—	46.4	—	—	—	—
Pasture-derived C, 10- to 20-cm	Single exponential decay	$F(t) = 24.1*exp(t* - 0.02)$	0.84	—	—	80.7	***	34.7	—	***
Bulk soil, 20- to 30-cm	Segmented	$F(t) = 19.91 - 0.23*t$	0.75	***	NS	75.08	—	—	—	—
OP-derived C, 20- to 30-cm	Linear	$F(t) = -0.20 + 0.07*t$	0.7	—	—	45.56	—	—	—	—
Pasture-derived C, 20- to 30-cm	Single exponential decay	$F(t) = 19.8*exp(t* - 0.02)$	0.79	—	—	77.7	***	40.8	—	***
Bulk soil, 30- to 50-cm	Linear	$F(t) = 29.42 - 0.25*t$	0.71	—	—	87.73	—	—	—	—
OP-derived C, 30- to 50-cm	Linear	$F(t) = 0.02 + 0.10*t$	0.60	—	—	66.39	—	—	—	—
Pasture-derived C, 30- to 50-cm	Single exponential decay	$F(t) = 29.8*exp(t* - 0.016)$	0.69	—	—	99	***	43.3	—	***

reaches  $112.3 \pm 3.2 \text{ Mg C ha}^{-1}$  (Fig. 3). This indicates that the conversion of pastures to OP plantations in this region is close to C neutrality ( $-0.5 \pm 8.8 \text{ Mg C ha}^{-1}$ ) considering multiple OP cycles. The rate of C accumulation in the biomass was higher ( $3.3 \pm 0.1 \text{ Mg C ha}^{-1} \text{ year}^{-1}$ ) than the rate of SOC losses ( $1.26 \pm 0.3 \text{ Mg C ha}^{-1} \text{ year}^{-1}$ ). As a result, TEC during the first rotation cycle was always higher than in pasture, but only about half of the time for the subsequent second rotation cycle when SOC stocks have reached equilibrium (Fig. 3). Therefore, OP plantations were continuously a C sink, even during the phase of SOC losses, except at the time of plantation replanting when OP biomass is destroyed.

**Soil fertility**

SOC content, unlike SOC stocks, increased significantly after  $41.1 \pm 2.7$  years following an initial decline in the 0- to 10-cm. However, this C recovery did not reach the initial soil C content present under pasture ( $2.5 \pm 0.1\%$ ; Fig. 4A). The rate of increase in soil C content was almost the double as compared to the decrease ( $-0.03 \pm 0.02\%$ ). This partial recovery of SOC content was observed down to a depth of 30 cm but not below where SOC content was still decreasing linearly ( $R^2 = 0.75$ ) after 56 years of OP cultivation.

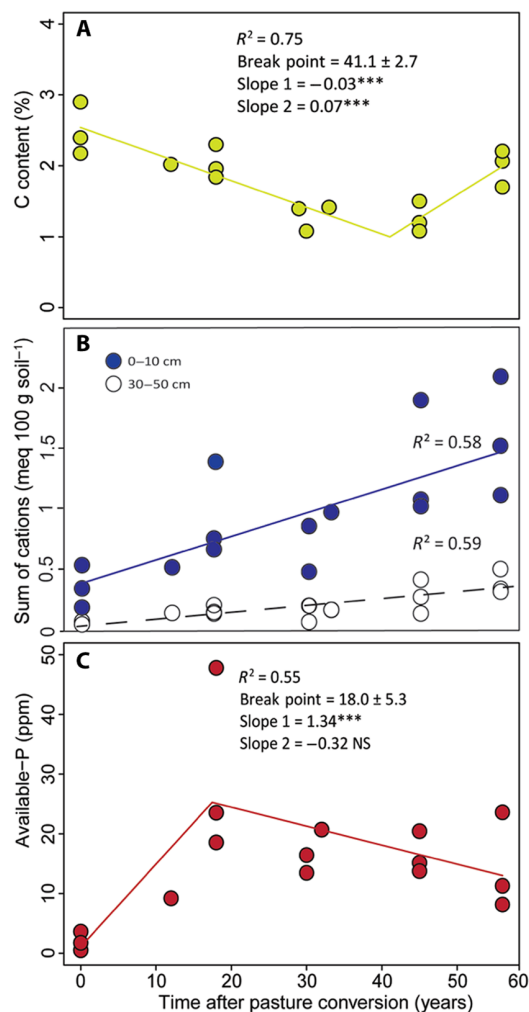
Macronutrient availability was strongly enhanced down to 50-cm depth (table S2). The sum of cations showed a decreasing trend with depth at all sites. Nonetheless, all layers exhibited a similar enrichment factor of 3.9 to 5.6 of the sum of cations, indicating nutrient leaching of the applied nutrients by fertilization. Nutrient enrichment did not show any saturation with time (Fig. 4B). Base saturation (BS) increased in the four layers, up to a factor of four, 56 years after conversion in the 30- to 50-cm-depth layer. Available Bray P showed a different pattern. It peaked at  $18.0 \pm 5.3$  years and then levelled off at higher levels than in pastures in the top 10 cm (Fig. 4C). This same pattern with small variations in the estimated break point was found for the 30- to 50-cm soil depth. However, a significant decrease during the last years of the first cycle and the entire second



**Fig. 3. TEC stocks.** TEC in OP plantations and pastures includes above- and belowground biomass and SOC stocks down to 50 cm (but not dead trees after replanted). The orange diamond and its vertical SE bars correspond to the time-averaged TEC stocks in pastures. Purple circle and its vertical SE bars correspond to time-averaged TEC stocks in OP, and purple band indicates the time-averaged TEC stocks during the 56 years of OP cultivation.

cycle followed the observed peak. This suggests the leaching of P into the subsoil in the short term (first OP cycle). Soil pH values exhibited a narrow range from 4.0 to 4.6 across land uses and soil depths.

OP cultivation did not have a constant effect throughout the two rotation cycles and soil depth, as shown by principal components analysis (PCA; Fig. 5). During the first cycle, the amount (SOC content) and quality (C/N ratio) of OM decreased, while the nutrients and bulk density (BD) increased in the surface (0- to 10-cm) and subsoil (30- to 50-cm) layers, as shown by the first principal component

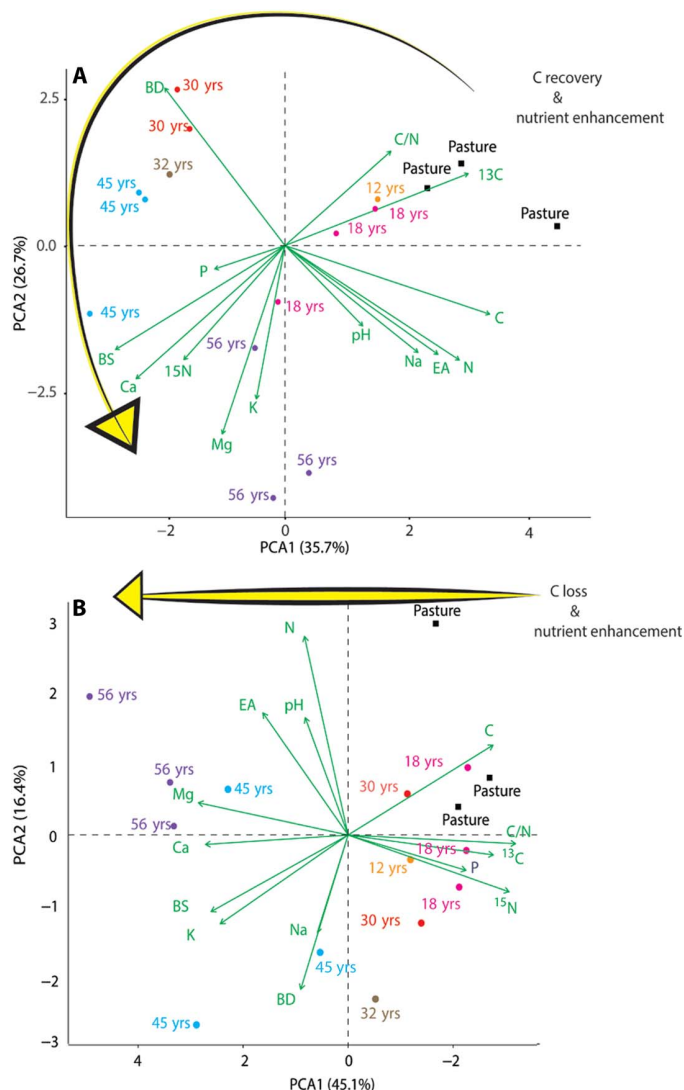


**Fig. 4. Soil chemical fertility dynamics.** (A) Soil C content as a function of time after pasture conversion into OP plantations at the soil surface (0- to 10-cm). The yellow line represents the fitted segmented regression equation. Significance of the slope coefficients from each side of the breaking point is indicated ( $***P < 0.001$ ). (B) Sum of cations as a function of time after pasture conversion into OP plantations at two soil depths: close circles, 0- to 10-cm; open circles, 30- to 50-cm. The lines represent the linear regression equations. (C) Available phosphorus (Bray P) as a function of time after pasture conversion into OP plantations at the soil surface (0- to 10-cm). The red line represents the fitted segmented regression equation. Significance of the slope coefficients from each side of the breaking point is indicated ( $***P < 0.001$ ).

(PC1, 35.7% at 0- to 10-cm and 45.1% at 30- to 50-cm). During the second cycle, the effects of cultivation remained similar in the subsoil (Fig. 5B), while the surface soil layer showed a partial recovery of SOC content but not SOC quality, suggesting a slight improvement of top-soil fertility in the oldest plantations.

**DISCUSSION**

We demonstrate that the conversion of pasture-to-OP in Colombia compares favorably in terms of ecosystem C changes to that of OP expansion into forested lands, as often occurs in major OP-producing countries in Southeast Asia. While the conversion of rainforests in Sumatra (6) led to a loss of 173 Mg C ha<sup>-1</sup>, the conversion



**Fig. 5. PCA of soil properties in pastures and OP plantations derived from pastures.** (A) The soil surface (0- to 10-cm) and (B) the subsoil (30- to 50-cm). BD, bulk density; C/N, C and N ratio; <sup>13</sup>C, δ<sup>13</sup>C; N, nitrogen; EA, exchangeable acidity; Na, sodium; K, potassium; Mg, magnesium; <sup>15</sup>N, δ<sup>15</sup>N; Ca, calcium; BS, base saturation and P, available phosphorus (Bray P). The OP plots are indicated with time after conversion in years (yrs).

of pasture-to-OP plantation was C neutral ( $-0.5 \pm 8.8 \text{ Mg C ha}^{-1}$ ), reducing TEC losses by  $99.7 \pm 9.6\%$ . In addition, establishing OP on pastures not only avoids the large initial loss of C stored in rainforest biomass but also increases biomass C pools. Nonetheless, converting pastures to OP did not act as C sink because the conversion caused large SOC reductions that counterbalanced the gains in OP biomass C.

The time-averaged C stored in OP biomass is mainly influenced by the length of rotation cycles (6). Shortening the rotation time to 25 years, as common in Southeast Asia, would reduce the time-averaged biomass C by 17% (about 8 Mg C ha<sup>-1</sup>), leading to a small C loss following land use change. OP biomass C found in Colombian plantations was similar to those reported for Indonesian's (6, 37). Carbon pools that were not measured in this study, such as frond piles or understory vegetation if present ( $<3 \text{ Mg C ha}^{-1}$ ), have a negligible contribution to TEC (37). The main factor determining whether

the land use change will result in small C gains or losses is the initial amount of SOC in the converted grassland ecosystem. SOC losses after land use change are not constant but rather proportional to the initial SOC stocks (20). The initial variability of SOC stocks in the studied pastures (SD of 14.3 Mg C ha<sup>-1</sup>) surpassed alone the C stored in grass biomass. The managed pastures investigated in this study could represent the land use type with the highest initial SOC storage on well-drained soils in this region, as *Brachiaria* pastures favor soil C sequestration when they replace either forested or grassland areas in the Neotropics (38, 39). In addition, because the rainfall and rainy season length are higher, soils in the piedmont often harbor slightly higher C stocks than the savannas further to the east. Lower SOC losses, and thus a slightly positive change in ecosystem C storage, are therefore expected when OP plantations are established on degraded pastures or other part of the Llanos (40). The same argument is often used to justify the transformation of the native savannas, without due regard to the biodiversity and ecosystem service values of these natural ecosystems. Given that the Llanos is expected to remain an important OP expansion area in Colombia, special attention needs to be given to the loss of native savannas as opposed to pastures.

The amount of SOC lost after the conversion of these pastures was higher than the ones reported when rainforests were converted to OP plantations, likely due to the high amount of SOC (102 Mg ha<sup>-1</sup>) stored down to 50 cm in the studied pastures (19, 20, 41). This finding is in line with those previously found in a Brazilian study, where SOC losses were more pronounced for OP plantations derived from pasture than from forested areas (17). However, the proportion of initial SOC lost in this study (39%) was similar to the proportion reported for rainforest-derived plantations, with the difference that it took longer to reach a new equilibrium in the pasture-derived plantations (20).

The dynamics of SOC stocks after land use change is commonly assumed to follow a single exponential decay until it reach a new equilibrium, i.e., most losses occur within a decade after conversion (20, 42, 43). In this study, losses were constant and extended over a longer time. The initial sharp decrease followed by a stabilization phase confirms the slow response of SOC to land use change and highlights the importance to assess changes in C cycling dynamics either over longer time scales or to investigate more sensitive SOC indicators, i.e., particulate OM, than total SOC pool size when only plantations within the first rotation cycle are studied. The dynamics of total SOC stocks was the net result of distinct patterns of C pool dynamics depending on soil depth and C origin. The decomposition of pasture-derived SOC did actually follow a single exponential decay in each layer, while OP-derived SOC accumulated at constant rate, except for 0- to 10-cm depth where SOC accumulation reached saturation. Despite the apparent stabilization of total SOC stocks down to 50 cm during the second rotation cycle, the isotopic approach demonstrated that only the top 10-cm layer was close to the equilibrium level (Fig. 2A). Between 10- and 30-cm depths, rates of pasture-derived C loss and OP-derived C stabilization were similar in the oldest plantations, so that bulk stocks appeared to have reached equilibrium. The decomposition rate in the 30- to 50-cm layer was still faster than the stabilization rate, explaining that the subsoil was still losing C after 56 years of OP cultivation. This finding is of relevance as no studies have reported, so far, effects of land use change to OP in subsoil horizons, presumably due to the relative short-term duration of the existent literature.

It is not possible yet to determine at which level SOC stocks will stabilize for layers below 10-cm depth since the accumulation of OP-derived C did not show any sign of saturation even after 56 years.

Under certain conditions, SOC stocks reach a minimum before increasing again, as the common U-shaped reported in other long-term SOC dynamics studies (44, 45). Because the rate of pasture-derived SOC losses decreased with time after conversion, stabilization rates can surpass decomposition rates if the stabilization does not saturate before. This mechanism can explain the recovery of C content in the top 10 cm (Fig. 4A) and would confirm the similar tendency observed for SOC stocks at least down to 20-cm depth (Fig. 2, A and B). It is commonly observed that SOC turnover slows down with increasing soil depth because of the higher level of SOC protection by minerals, reduced microbial activity, or reduced root C inputs (46). Because of slower C dynamics in deep soil, it is possible that layers below 10-cm depth will reach the recovery phase observed in the topsoil in the third rotation cycle. Nonetheless, it is questionable whether the soil below 30-cm depth will follow the same dynamics as the upper layer to eventually reach this stage (Fig. 5B). OP rooting system has a specific architecture with fine roots growing upward to the surface from exploratory coarse roots growing horizontally at around 30-cm depth (47). Hence, root biomass and activity are greatest in the top 30 cm. Layers below 30 cm receive less C inputs from roots but are still strongly enriched with nutrients leached from the heavy fertilization, favoring SOC mineralization. Consequently, despite the stabilization of SOC stocks during the second rotation cycle, stocks might slightly increase or decrease in subsequent rotation cycles depending on the intensity of the recovery in the upper layer and the stabilization level below 30-cm depth.

Our findings demonstrate that the conversion of pasture into OP had an important impact on SOC stocks. The most important long-term changes in SOC storage occurred during the first OP rotation cycle where 39% of the original SOC was lost. The second phase was characterized by a redistribution of SOC within the soil profile, in which a recovery of SOC in the topsoil compensates for a decrease in the subsoil, leading to a stabilization of the total SOC stocks down to 50-cm depth. The negative impact of OP on SOC storage was high when cultivated on pastures, but it was comparable to the impact of intensive cultivation of other nonperennial crops on pasture or rainforests (16, 26). Soil degradation was mainly limited to a decrease of SOM. The typical threats affecting soil physical quality in OP plantations established on rainforest such as erosion and compaction were limited in the studied plantations (19, 48). Although the reference pasture soils might have been compacted by trampling, native savanna soils in the Llanos region are naturally compacted and thus less sensitive to further compaction (49). In addition, the topography of the Llanos region is mostly composed of flat areas (i.e., piedmont and flat high plains) reducing the risk of soil erosion, contrasting with Indonesian islands where plantations are affected by severe soil erosion (19, 50).

Because of the soil preparation before OP establishment and frequent fertilizer applications, inherent with low soil quality of the local soils, soil physical and chemical fertility remained similar or even higher after two OP cultivation cycles. Specifically, while SOC declined, the sum of exchangeable bases increased during the pastures to OP transition, implying a trade-off between soil chemical properties and SOC in the long term. This suggests that OP nutrient supply relies mostly on the frequent application of mineral fertilizers and not on nutrients released from SOM mineralization by microorganisms. Nonetheless, soils did not show signs of degradation that would prevent the establishment of new crops or the restoration of (semi-) natural plant communities following OP cultivation. This is fundamental because TEC changes remain theoretical since it assumes that

OP cultivation will last permanently. Ultimately, the impact of land use change will depend on the land use following OP. Since biomass C stock gains disappear rapidly while SOC losses remain, the following land use type would start with a C debt of about 40 Mg C ha<sup>-1</sup> in the soil after 60 years of OP cultivation. It is, therefore, important to maintain soil physical and chemical properties because SOC losses from land use change are, to a high degree, reversible (26). The sustainability of OP cultivation would, therefore, benefit from the implementation of management practices that incorporate organic residues, either as empty fruit bunches, compost, or by using cover crops into the regular soil management programs.

Soils under OP plantations in this study were far from C saturation, as indicated by the lower SOC stocks than those found in pastures and by the linear increase of OP-derived C observed in subsoil layers. Therefore, a large potential for C sequestration in soils subjected to this land use change alternative can be expected if OM inputs increase. Furthermore, increasing SOC and nutrients in an organic form limits the trade-off that negatively affects soil biota by increasing synergies between soil chemical properties and SOC (51, 52). Replacing soil biological activity by mineral fertilization is risky if fertilizer costs increase or if their supply decreases in the future. Moreover, soil biota improve soil stability and resilience, facilitating the restoration of former OP plantations (53).

Pastures tend to emit more nitrous oxide (N<sub>2</sub>O) emissions than tropical forests (54); therefore, if OP will expand on pastures, then it is plausible that this alternative land use change will compare positively to the common deforestation scenario in terms of N<sub>2</sub>O emissions. Adoption of a suite of practices for better nutrient management, i.e., customized fertilization programs, should be encouraged in OP plantations not only to reduce mineral nutrient losses but also to limit their stimulating effect on SOM decomposition in subsoil layers, favoring deep SOC depletion, and reduce the greenhouse gas emission budget from preventing N<sub>2</sub>O emissions.

OP is blamed for its large environmental impacts, especially the reduction of C stocks and biodiversity from tropical forest conversion, so the search for low-impact land use change alternatives has become imperative. This study provides empirical field-based evidence that the conversion of pastures to OP is C neutral from an ecosystem C storage perspective. The availability of large pasture areas in the tropics, particularly in Latin America, could limit the negative impacts of ongoing OP expansion to native savannas and natural forests (2, 14). Our findings indicate that conversion of pastures to OP can be an opportunity to preserve and even increase C sequestration in the tropics and reduce the large C footprint of OP development on forested land. Recent studies on biodiversity in the Neotropics have also demonstrated that OP plantations established on pastures can spare the development of other ecosystems that are much richer in endemic and threatened vertebrates (55, 56).

The spatial design of OP plantations can also take greater account of other natural land cover elements, including remnant forests and savannas. This more heterogeneous landscape mosaic would likely provide increased ecosystem resilience favoring both productivity and conservation. We hope that our study will prompt research on other fundamental aspects of OP expansion in pastures areas, such as seasonal water scarcity and socioeconomic aspects. We recognized that a shift in OP expansion from forested land to unproductive pastures will need to be supported by policy. Improving support services, including infrastructure development such as mills and transport facilities and facilities for the substantial labor requirements that will be

required, could encourage the development of OP in regions where there are unproductive pastures. Further measures like banning forest conversion or, on the consumer side, demanding effective certification schemes for OP production would also reduce deforestation for OP expansion and its marked environmental impacts.

## MATERIALS AND METHODS

### Study area

The study was carried out in “La Cabaña,” a large-scale commercial OP plantation (73°22'W, 4°16'N) and three adjacent cattle ranching farms, close to the town of Cumaral, of Meta Department in Colombia. The area is located in the piedmont of the Llanos region in the eastern plains, close to the Andean mountains, with an altitude of 300 m (fig. S1). The climate in the area is tropical with a well-marked dry season that lasts from December through March. Annual rainfall is of about 3400 mm, and annual mean temperature is approximately 27°C. The study area is located on the well-drained Pleistocene and late Tertiary alluvial terraces, where soils are predominantly dystrophic Inceptisols (Oxic Dystropepts) of about 60- to 70-cm effective depth overlying coarse alluvial sediments on flat topography.

In general, the Llanos is a vast territory mostly dominated by a mosaic of savanna (C4 dominated) and gallery forest ecosystems that, over the past few decades, has undergone fast land use change to intensive commercial agriculture and is often regarded as one of the last frontiers for agricultural expansion in South America (34, 57). The region accounts for about one-fourth of the national territory, respectively 22 million ha (34, 58). Cattle ranching is the predominant land use in the piedmont and of the Llanos in general, where large areas of cleared forests and savannas have been sown with improved varieties of *Brachiaria* grasses for several decades (*Brachiaria* spp.) (59). *Brachiaria* grasses are of African origin, and they are widely used in improved South-American C4 pasture systems. However, during the last 40 years, increasing areas of these pastures and natural savannas have transitioned to intensive agriculture of rice and OP plantations. Because of the economic and social benefits, a suit of favorable climatic conditions in the Andean piedmont of the Llanos and increasingly governmental stimulus, OP plantations are being established since more than half a century, and their expansion is predicted to continue (32). As is the case for most OP plantations in Colombia, the study site plantations at La Cabaña were derived from pastures, which had been planted on former well-drained grassland savannas.

### Approach and study sites

We used a space for time substitution approach (chronosequence) to quantify the long-term impacts on biomass and soil properties, including changes in SOC stocks and soil chemical fertility characteristics, following pasture conversion into OP plantations. As we aimed to study the long-term effects of OP cultivation on soil properties, OP plots with palm stands passing by the first rotation cycle (up to 30 years) and the second rotation cycle (new palms transplanted after the first cycle) were considered. We selected six OP plots that range from 12 to 56 years after pasture conversion and three reference pasture sites. All OP plots were part of a large-scale OP plantation. This large-scale OP farm could have the longest history of OP cultivation in the whole Llanos region, and its management is representative of the typical management of OP plantations in this region of Colombia. The reference sites were cattle ranching farms that were adjacent to the sampled OP farm. The primary criterion for selecting OP plots was

the existence of pastures as the sole preceding land-use. This information was obtained by direct communication with the personnel in charge of both the OP farm operation and the neighboring cattle ranching farms. In addition, in the plot selection process, sites with steep slopes, inundated parts, or located on floodplains and with distinct management to the general farm practices were excluded. OP plots were planted in a triangular design 9 m apart, which resulted in a density of 143 palms ha<sup>-1</sup>. All OP plots, except the recently replanted 32-year plot, presented four well distinct management zones: (i) the frond piles located in between palms and where pruned OP fronds are accumulated; (ii) the harvest path, which is the area of traffic for mechanized operation, i.e., fruit harvesting, in between parallel palm lines; (iii) weeded circle, which is an area around the palm trunk of about 5 m in diameter where most fertilizer inputs are placed up until 5 to 6 years; and (iv) the interrow where almost no field operations takes place and scattered understory vegetation, i.e., weeds, grows. Land was prepared prior plantation establishment by chisel plow to a depth of 10 to 15 cm. Dolomitic lime was added to increase soil pH. Fertilization practices were the typically recommended ones for the region and were done periodically every year in two applications on the weeded circle at young ages or broadcasted all over the plantation except the harvest path in mature plantations (over 5-year plantations). Nitrogen (N), phosphorus (P), and potassium (K) fertilization of 150 to 725 kg ha<sup>-1</sup> year<sup>-1</sup> were applied depending on palm stand age, i.e., 725-kg NPK-complete fertilizer applied to plantation aging more than 10 years. Other nutrients including boron and magnesium were also applied regularly every year.

### Site selection and sampling

Soil samples were taken from a chronosequence of OP with stand ages of 12, 18, 30, 32, 45, and 56 years, the three first blocks correspond to first cycle plantations and the last three to second cycle plantations. OP blocks were of sizes between 20 and 30 ha. In addition, three adjacent cattle ranching farms were sampled as reference sites. All selected sites were located within an area that covered roughly 5000 ha.

Soil samples from OP plots were collected using a modified transect methodology (60). This sampling strategy allows us to have mixed samples that represent well the spatial variability in OP plantations due to management practices. Furthermore, it is also well adapted for measures in replanted plantations, since the spatial distribution of management zones differs between OP cycles. Twenty sampling points evenly spaced were marked along a 50-m diagonal transect at ca. >60° (considering a selected palm at row 1 of the transect as reference point). The diagonal transects crossed six OP rows, and the four management zones were typically found in commercial OP plantations. To capture the spatial variability at each of the OP blocks, three parallel transects were made, except in the 12- and 32-year OP blocks, and sampled in 2016, with some complementary sampling in 2017. One transect was established in a centered position relative to the number of palm rows and average number of palms per row in each block. The other two transects were established at least 120 m away from each side of the first transect. Transects were established at least 50 m away from the plot's edges. At the three *Brachiaria* pasture sites, one diagonal transect of 50 m long oriented east-west with 20 evenly spaced sampling points was made. At all diagonal transects (OP blocks and pasture sites), a total of 20 soil cores were taken. Half of those cores were taken to a depth of 30 cm and the other half went down to 50 cm in an alternative manner, using a 6-cm-diameter soil

auger. Cores were divided into four depth intervals: 0 to 10 cm, 10 to 20 cm, 20 to 30 cm, and 30 to 50 cm. Therefore, samples of the 0 to 10 cm, 10 to 20 cm, and 20 to 30 cm were composed of 20 cores, and the 30- to 50-cm samples were made of 10 cores. Final samples in each transect were bulked into a composite sample, resulting in one sample for each depth in each transect. Samples were homogenized, air-dried, sieved through 2 mm, sealed in bags, and stored at room temperature until transportation to the laboratory in Lausanne (Switzerland). Soil physical, chemical, and isotopic analyses were carried out after drying at 35°C in forced-air ovens.

Pits were dug in a middle representative position along the first 50-m linear transects to determine soil BD in each OP block and one reference site to 70-cm depth. Two stainless steel volume cores were inserted horizontally into a pit wall at each depth increment. Soils from the two cores were mixed and oven-dried at 105°C for 48 hours to calculate BD.

### Above- and belowground biomass

At the previously described central diagonal transects of each OP sampled plot, the heights of 10 randomly chosen palms were measured. Palm height was measured from the palm base to the base of the youngest fully expanded leaf (61). Estimation of aboveground biomass was based on palm height using the allometric equation for mineral soils (37)

$$\text{AGB OP} = 0.0923(\text{height}) + 0.1333 \quad (1)$$

Belowground OP biomass was estimated according to the allometric model based on OP plantation age (Eq. 2) (62)

$$\text{BGB OP} = 1.45 (\text{age}) + 9.88 (143 \text{ palms ha}^{-1}) \quad (2)$$

Biomass C stocks were estimated using a factor of 41.3% of above- and belowground OP biomass (62). Time-averaged C in OP biomass was estimated as the C stocks accumulated in the middle of one rotation cycle; thus, the biomass C stock in a 30-year-old OP plantation was divided by two (63).

### Laboratory analyses

Soil particle size analysis was performed on air-dried soils by the hydrometer method after removal of organic fraction with 30% H<sub>2</sub>O<sub>2</sub> (64). Soil pH was determined in a 1:2.5 soil-to-water slurry. Soil samples were extracted with Mehlich-III solution (65) and analyzed for available cations including Ca, K, Na, and Mg using an inductively coupled plasma spectrometer (PerkinElmer, Waltham, MA, USA). Exchangeable acidity was determined by extracting 2 g of soil with 10 ml of 1N KCl, shaking for 30 min at 200 rpm. Samples were allowed to settle for 30 min, and filtered, and extraction funnels were washed three times with 30 ml of 1N KCl. Titration was conducted with 0.01N NaOH after adding phenolphthalein to the extract (66). Effective cation exchange capacity (ECEC) was estimated by summing the amount of charge per unit soil (meq 100 g<sup>-1</sup>) from major cations (Ca, K, Na, and Mg) plus exchangeable acidity. BS was obtained by dividing the total sum of charge per unit soil from Ca, K, Na, and Mg by ECEC.

Total C and N contents, <sup>13</sup>C, and <sup>15</sup>N were measured on air-dried and ground soil weighted in tin cups with an Elemental Analyser (EuroVector) coupled to an isotope ratio mass spectrometer (Delta



plus, Thermo Fisher Scientific) at the stable isotope lab at the Swiss Federal Institute for Forest, Snow and Landscape Research WSL, Birmensdorf, Switzerland. Total C content corresponds to total organic content because of the absence of carbonates in heavily weathered soils.

The SOC stocks were calculated as the product of soil BD, layer thickness, and SOC concentration. The contribution of new and old C to total SOC was calculated as shown in Eq. 3 (67)

$$f = (\delta_{\text{sam}} - \delta_{\text{ref}}) / (\delta_{\text{op}} - \delta_{\text{ref}}) \quad (3)$$

where  $f$  is the relative proportion of OP-derived C (C3) in SOC stocks.  $\delta_{\text{sam}}$  is the  $\delta^{13}\text{C}$  of the soil sample, and  $\delta_{\text{ref}}$  is the  $\delta^{13}\text{C}$  of the corresponding soil depth from pasture as C4 reference soil.  $\delta_{\text{op}}$  is the  $\delta^{13}\text{C}$  of nine fine root biomass C samples. The method accounts for the natural increase of  $\delta^{13}\text{C}$  signature with depth generally observed in soils and neglects the  $^{13}\text{C}$  fractionation that could occur at the first step of SOC formation.

### Statistical analyses

Data analyses were performed using RStudio version 3.4.0 (R Development Core Team 2017) statistical software. Linear regression analysis was used to examine changes in soil chemical and physical properties (i.e., major cations and BD) relative to increase time after pasture conversion into OP plantations. Assumption of normality was checked for all analyses with Shapiro-Wilk test and visual inspection of normality plots. If assumption of normality was not satisfied, then permutation tests were performed as in the case of Ca, sum of cations, and BS. Patterns in changes of soil C stocks over the OP chronosequence time were examined for the bulk soil and for each of the four sampled soil layers using regression models. The nonlinear least square “nlm” and linear model “lm” functions in R were used to fit nonlinear and linear regression models, respectively. In addition, the “segmented” function was used to perform segmented (broken line) regression analysis. Statistical significance was declared at  $P < 0.05$ . Similarly, model fit for changes in C3-OP-derived C and C4 pasture-derived C over the chronosequence time was also examined for each of the four studied soil layers by testing the above-mentioned models. These regression analyses allowed estimating rates of C3 accumulation and C4 decomposition and decay constants ( $k$ ), rates of total soil C decrease, and break points in soil C stock changes (time at which a change in the direction of change in C stocks occurred).

After testing various models (i.e., mono-exponential and bi-exponential), model performance assessment was based on Akaike information criterion (AIC) and coefficient of determination ( $R^2$ ) values. Models with the largest  $R^2$  and lowest AIC values were selected (see table S1). Accordingly, (i) changes in whole soil C stocks and C stocks in each soil layer were described by fitting segmented regression models (except linear regression on the deepest soil layer of 30 to 50 cm), which yielded estimated break points; (ii) C3-derived C in the 10- to 20-cm, 20- to 30-cm, and 30- to 50-cm soil layers were described by linear models, while pattern in C3-derived C in the soil surface layer (0 to 10 cm) was described by an exponential rise to equilibrium model of the form

$$y = ((k * y_0 - A) * \exp(-k * t + A)) / k \quad (4)$$

where  $y$  is the C stock,  $k$  is the annual decay constant of the pool,  $y_0$  is the C3-C stock before OP cultivation started (thus, 0),  $A$  is the C3

annual input to the C3 pool, and  $t$  is time after conversion; (iii) A first-order decay model was fitted to the obtained pasture-derived C data in the four studied soil layers

$$y = c * \exp(-k * t) \quad (5)$$

where  $y$  is the C stock,  $c$  is the original C stock before OP cultivation,  $k$  is the decay rate constant, and  $t$  referred to time. The half-life (HL) of the original C stock in Eq. 5 and of the OP input in Eq. 4 was calculated as

$$\text{HL} = \ln(2) / k \quad (6)$$

A PCA was carried out for further exploration of the relations between soil parameters that can be affected by cultivation (BD, C:N,  $^{13}\text{C}$ , C, N, EA, Na, pH, K, Mg,  $^{15}\text{N}$ , Ca, BS, and P).

### SUPPLEMENTARY MATERIALS

Supplementary material for this article is available at <http://advances.sciencemag.org/cgi/content/full/5/11/eaaw4418/DC1>

Fig. S1. Map of study sites and the potential area for the expansion of OP in pasture lands in the Neotropics.

Table S1. Set of models tested for the bulk soil and pasture- and OP-derived carbon.

Table S2. Soil chemical, physical, and isotopic properties.

### REFERENCES AND NOTES

1. D. L. A. Gaveau, D. Sheil, Husnayaen, M. A. Salim, S. Arjasakusuma, M. Ancrenaz, P. Pacheco, E. Meijaard, Rapid conversions and avoided deforestation: Examining four decades of industrial plantation expansion in Borneo. *Sci. Rep.* **6**, 32017 (2016).
2. V. Vijay, S. L. Pimm, C. N. Jenkins, S. J. Smith, The impacts of oil palm on recent deforestation and biodiversity loss. *PLOS ONE* **11**, e0159668 (2016).
3. E. B. Fitzherbert, M. J. Struebig, A. Morel, F. Danielsen, C. A. Brühl, P. F. Donald, B. Phalan, How will oil palm expansion affect biodiversity? *Trends Ecol. Evol.* **23**, 538–545 (2008).
4. C. Dislich, A. C. Keyel, J. Salecker, Y. Kisel, K. M. Meyer, M. Auliya, A. D. Barnes, M. D. Corre, K. Darras, H. Faust, B. Hess, S. Klasen, A. Knohl, H. Kreft, A. Meijide, F. Nurdiansyah, F. Otten, G. Pe'er, S. Steinebach, S. Tarigan, M. H. Tölle, T. Tschardtke, K. Wiegand, A review of the ecosystem functions in oil palm plantations, using forests as a reference system. *Biol. Rev.* **92**, 1539–1569 (2017).
5. S. Henders, U. M. Persson, T. Kastner, Trading forests: Land-use change and carbon emissions embodied in production and exports of forest-risk commodities. *Environ. Res. Lett.* **10**, 125012 (2015).
6. T. Guillaume, M. M. Kotowska, D. Hertel, A. Knohl, V. Krashevskaya, K. Murtlaksono, S. Scheu, Y. Kuzyakov, Carbon costs and benefits of Indonesian rainforest conversion to plantations. *Nat. Commun.* **9**, 2388 (2018).
7. D. Boucher, in *Tropical Forest and Climate Initiative* (Union of Concerned Scientists, 2011), pp. 1–5.
8. J. Lucey, J. Hill, P. van der Meer, G. Reynolds, F. Agus, Change in carbon stocks arising from land-use conversion to oil palm plantations (2014).
9. L. P. Koh, J. Miettinen, S. C. Liew, J. Ghazoul, Remotely sensed evidence of tropical peatland conversion to oil palm. *Proc. Natl. Acad. Sci. U.S.A.* **108**, 5127–5132 (2011).
10. L. P. Koh, J. Ghazoul, Spatially explicit scenario analysis for reconciling agricultural expansion, forest protection, and carbon conservation in Indonesia. *Proc. Natl. Acad. Sci. U.S.A.* **107**, 11140–11144 (2010).
11. J. Garcia-Ulloa, S. Sloan, P. Pacheco, J. Ghazoul, L. P. Koh, Lowering environmental costs of oil-palm expansion in Colombia. *Conserv. Lett.* **5**, 366–375 (2012).
12. A. D. Ziegler, J. Phelps, J. Q. Yuen, E. L. Webb, D. Lawrence, J. M. Fox, T. B. Bruun, S. J. Leisz, C. M. Ryan, W. Dressler, O. Mertz, U. Pascual, C. Padoch, L. P. Koh, Carbon outcomes of major land-cover transitions in SE Asia: Great uncertainties and REDD+ policy implications. *Glob. Chang. Biol.* **18**, 3087–3099 (2012).
13. J. J. Gilroy, G. W. Prescott, J. S. Cardenas, P. González del Pliego Castañeda, A. Sánchez, L. E. Rojas-Murcia, C. A. Medina Uribe, T. Haugaasen, D. P. Edwards, Minimizing the biodiversity impact of Neotropical oil palm development. *Glob. Chang. Biol.* **21**, 1531–1540 (2015).
14. P. R. Furumo, T. M. Aide, Characterizing commercial oil palm expansion in Latin America: Land use change and trade. *Environ. Res. Lett.* **12**, 024008 (2017).
15. D. Li, S. Niu, Y. Luo, Global patterns of the dynamics of soil carbon and nitrogen stocks following afforestation: A meta-analysis. *New Phytol.* **195**, 172–181 (2012).

16. L. B. Guo, R. M. Gifford, Soil carbon stocks and land use change: A meta analysis. *Glob. Chang. Biol.* **8**, 345–360 (2002).
17. L. A. Frazão, K. Paustian, C. E. Pellegrino Cerri, C. C. Cerri, Soil carbon stocks and changes after oil palm introduction in the Brazilian Amazon. *GCB Bioenergy* **5**, 384–390 (2013).
18. I. Goodrick, P. N. Nelson, M. Banabas, C. M. Wurster, M. I. Bird, Soil carbon balance following conversion of grassland to oil palm. *GCB Bioenergy* **7**, 263–272 (2015).
19. T. Guillaume, M. Damris, Y. Kuzuyakov, Losses of soil carbon by converting tropical forest to plantations: Erosion and decomposition estimated by  $\delta^{13}\text{C}$ . *Glob. Chang. Biol.* **21**, 3548–3560 (2015).
20. O. van Straaten, M. D. Corre, K. Wolf, M. Tchienkoua, E. Cuellar, R. B. Matthews, E. Veldkamp, Conversion of lowland tropical forests to tree cash crop plantations loses up to one-half of stored soil organic carbon. *Proc. Natl. Acad. Sci. U.S.A.* **112**, 9956–9960 (2015).
21. N. Khasanah, M. van Noordwijk, H. Ningsih, S. Rahayu, Carbon neutral? No change in mineral soil carbon stock under oil palm plantations derived from forest or non-forest in Indonesia. *Agric. Ecosyst. Environ.* **211**, 195–206 (2015).
22. D. R. Smith, T. J. Townsend, A. W. K. Choy, I. C. W. Hardy, S. Sjögersten, Short-term soil carbon sink potential of oil palm plantations. *GCB Bioenergy* **4**, 588–596 (2012).
23. K. Allen, M. D. Corre, A. Tjoa, E. Veldkamp, Soil nitrogen-cycling responses to conversion of lowland forests to oil palm and rubber plantations in Sumatra, Indonesia. *PLOS ONE* **10**, e0133325 (2015).
24. T. G. Bárceña, L. P. Kiær, L. Vesterdal, H. M. Stefánsdóttir, P. Gundersen, B. D. Sigurdsson, Soil carbon stock change following afforestation in Northern Europe: A meta-analysis. *Glob. Chang. Biol.* **20**, 2393–2405 (2014).
25. K. I. Paul, P. J. Polglase, J. G. Nyakuengama, P. K. Khanna, Change in soil carbon following afforestation. *For. Ecol. Manage.* **168**, 241–257 (2002).
26. A. Don, J. Schumacher, A. Freibauer, Impact of tropical land-use change on soil organic carbon stocks - a meta-analysis. *Glob. Chang. Biol.* **17**, 1658–1670 (2011).
27. N. Fierer, A. S. Allen, J. P. Schimel, P. A. Holden, Controls on microbial  $\text{CO}_2$  production: A comparison of surface and subsurface soil horizons. *Glob. Chang. Biol.* **9**, 1322–1332 (2003).
28. S. Fontaine, S. Barot, P. Barré, N. Bdioui, B. Mary, C. Rumpel, Stability of organic carbon in deep soil layers controlled by fresh carbon supply. *Nature* **450**, 277–280 (2007).
29. E. Medina, Physiological ecology of neotropical savanna plants, in *Ecology of Tropical Savannas* B. J. Huntley, B. H. Walker, Eds. (Springer, 1982), pp. 308–335.
30. J. Balesdent, A. Mariotti, B. Guillet, Natural  $^{13}\text{C}$  abundance as a tracer for studies of soil organic matter dynamics. *Soil Biol. Biochem.* **19**, 25–30 (1987).
31. C. Castiblanco, A. Etter, T. M. Aide, Oil palm plantations in Colombia: A model of future expansion. *Environ. Sci. Policy* **27**, 172–183 (2013).
32. A. Rueda-Zárate, P. Pacheco, *Políticas, Mercados y Modelos de Producción* (CIFOR, 2015).
33. Fedepalma, Balance económico del sector palmero colombiano en 2014, *Boletín Económico* (2015).
34. E. Guimarães, *Agropastoral Systems for the Tropical Savannas of Latin America* (CIAT, 2004).
35. S. P. Braz, S. Urquiaga, B. J. R. Alves, C. P. Jantalia, A. Paula, P. Guimarães, C. A. dos Santos, S. C. dos Santos, É. F. Machado Pinheiro, R. M. Boddey, Soil carbon stocks under productive and degraded *Brachiaria* pastures in the Brazilian cerrado. *Soil Sci. Soc. Am. J.* **77**, 914–928 (2013).
36. O. Mosquera, P. Buurman, B. L. Ramirez, M. C. Amezcua, Carbon stocks and dynamics under improved tropical pasture and silvopastoral systems in Colombian Amazonia. *Geoderma* **189–190**, 81–86 (2012).
37. N. Khasanah, M. van Noordwijk, H. Ningsih, S. Wich, Aboveground carbon stocks in oil palm plantations and the threshold for carbon-neutral vegetation conversion on mineral soils. *Cogent Environ. Sci.* **1**, 1119964 (2015).
38. C. E. P. Cerri, M. Easter, K. Paustian, K. Killian, K. Coleman, M. Bernoux, P. Falloon, D. S. Powlson, N. H. Batjes, E. Milne, C. C. Cerri, Predicted soil organic carbon stocks and changes in the Brazilian Amazon between 2000 and 2030. *Agric. Ecosyst. Environ.* **122**, 58–72 (2007).
39. M. J. Fisher, S. P. Braz, R. S. M. Dos, S. Urquiaga, B. J. R. Alves, R. M. Boddey, Another dimension to grazing systems: Soil carbon. *Trop. Grasslands* **41**, 65–83 (2007).
40. J. J. Jiménez, R. Lal, Mechanisms of C sequestration in soils of Latin America. *Crit. Rev. Plant Sci.* **25**, 337–365 (2006).
41. T. Chiti, E. Grieco, L. Perugini, A. Rey, R. Valentini, Effect of the replacement of tropical forests with tree plantations on soil organic carbon levels in the Jomoro district, Ghana. *Plant and Soil* **375**, 47–59 (2014).
42. M. de Blécourt, R. Brumme, J. Xu, M. D. Corre, E. Veldkamp, Soil carbon stocks decrease following conversion of secondary forests to rubber (*Hevea brasiliensis*) plantations. *PLOS ONE* **8**, e69357 (2013).
43. I. Lobe, W. Amelung, C. C. Du Preez, Losses of carbon and nitrogen with prolonged arable cropping from sandy soils of the South African Highveld. *Eur. J. Soil Sci.* **52**, 93–101 (2001).
44. W. W. Covington, Changes in Forest Floor Organic Matter and Nutrient Content Following Clear Cutting in Northern Hardwoods. *Ecology* **62**, 41–48 (2011).
45. D. R. Zak, D. F. Grigal, S. Gleeson, D. Tilman, Carbon and nitrogen cycling during old-field succession: Constraints on plant and microbial biomass. *Biochemistry* **11**, 111–129 (1990).
46. J. Balesdent, I. Basile-Doelsch, J. Chadoeuf, S. Cornu, D. Derrien, Z. Fekiacova, C. Hatté, Atmosphere–soil carbon transfer as a function of soil depth. *Nature* **559**, 599–602 (2018).
47. C. Jourdan, H. Rey, Modelling and simulation of the architecture and development of the oil-palm (*Elaeis guineensis* Jacq.) root system. *Plant and Soil* **190**, 217–233 (1997).
48. T. Guillaume, D. Marangui, K. Murtillaksono, Y. Kuzyakov, Sensitivity and resistance of soil fertility indicators to land-use changes: New concept and examples from conversion of Indonesian rainforest to plantations. *Ecol. Indic.* **67**, 49–57 (2016).
49. P. Lavelle, N. Rodríguez, O. Arguello, J. Bernal, C. Botero, P. Chaparro, Y. Gómez, A. Gutiérrez, M. del Pilar Hurtado, S. Loaiza, S. X. Pullido, E. Rodríguez, C. Sanabria, E. Velásquez, Soil ecosystem services and land use in the rapidly changing Orinoco river basin of Colombia. *Agric. Ecosyst. Environ.* **185**, 106–117 (2014).
50. M. Gharibreza, J. K. Raj, I. Yusoff, Z. Othman, W. Z. W. M. Tahir, M. A. Ashraf, Land use changes and soil redistribution estimation using  $^{137}\text{Cs}$  in the tropical Bera Lake catchment, Malaysia. *Soil Tillage Res.* **131**, 1–10 (2013).
51. H.-H. Tao, E. M. Slade, K. J. Willis, J.-P. Caliman, J. L. Snaddon, Effects of soil management practices on soil fauna feeding activity in an Indonesian oil palm plantation. *Agric. Ecosyst. Environ.* **218**, 133–140 (2016).
52. H.-H. Tao, J. L. Snaddon, E. M. Slade, J.-P. Caliman, R. H. Widodo, Suhardi, K. J. Willis, Long-term crop residue application maintains oil palm yield and temporal stability of production. *Agron. Sustain. Dev.* **37**, 33 (2017).
53. S. F. Bender, C. Wagg, M. G. A. van der Heijden, An underground revolution: Biodiversity and soil ecological engineering for agricultural sustainability. *Trends Ecol. Evol.* **31**, 440–452 (2016).
54. J. Van Lent, K. Hergoualc'h, L. V. Verchot, Reviews and syntheses: Soil  $\text{N}_2\text{O}$  and  $\text{NO}$  emissions from land use and land-use change in the tropics and subtropics: a meta-analysis. *Biogeosciences* **12**, 7299–7313 (2015).
55. N. Ocampo-Peñuela, J. Garcia-Ulloa, J. Ghazoul, A. Etter, Quantifying impacts of oil palm expansion on Colombia's threatened biodiversity. *Biol. Conserv.* **224**, 117–121 (2018).
56. G. W. Prescott, J. J. Gilroy, T. Haugaasen, C. A. Medina Uribe, W. A. Foster, D. P. Edwards, Reducing the impacts of Neotropical oil palm development on functional diversity. *Biol. Conserv.* **197**, 139–145 (2016).
57. R. Lal, Soil carbon sequestration impacts on global climate change and food security. *Science* **304**, 1623–1627 (2004).
58. M. Romero-Ruiz, A. Etter, A. Sarmiento, K. Tansey, Spatial and temporal variability of fires in relation to ecosystems, land tenure and rainfall in savannas of northern South America. *Glob. Chang. Biol.* **16**, 2013–2023 (2010).
59. A. Etter, A. Sarmiento, M. Romero, Land Use Changes (1970–2020) and Carbon Emissions in the Colombian Llanos, in *Ecosystem Function in Savannas*, M. J. Hill, N. P. Hanan, Eds. (Taylor & Francis Group, 2010), pp. 383–402.
60. P. N. Nelson, M. Banabas, I. Goodrick, M. J. Webb, N. I. Huth, D. O'Grady, Soil sampling in oil palm plantations: A practical design that accounts for lateral variability at the tree scale. *Plant and Soil* **394**, 421–429 (2015).
61. M. M. Kotowska, C. Leuschner, T. Triadiati, S. Meriem, D. Hertel, Quantifying above- and belowground biomass carbon loss with forest conversion in tropical lowlands of Sumatra (Indonesia). *Glob. Chang. Biol.* **21**, 3620–3634 (2015).
62. C. V. Göttingen, Syahrudin, The potential of oil palm and forest plantations for carbon sequestration on degraded land in Indonesia (Cuvillier Verlag, 2005), *Ecology and Development Series*, vol. **28**, p. 115.
63. J. Penman, M. Gytarsky, T. Hiraishi, T. Krug, D. Kruger, R. Pipatti, L. Buendia, K. Miwa, T. Ngara, K. Tanabe, F. Wagner, *Intergovernmental Panel on Climate Change Good Practice Guidance for Land Use, Land-Use Change and Forestry Edited by* (2003); www.ipcc-nggip.iges.or.jp/public/gpplulucf/gpplulucf\_files/GPG\_LULUCF\_FULL.pdf, vol. 177.
64. G. W. Gee, J. W. Bauder, Particle-size analysis, in *Methods of Soil Analysis, Part 1. Physical and Mineralogical Methods, Agronomy Monograph No. 9, 2nd Edition* (American Society of Agronomy/Soil Science Society of America, Madison, WI 1986), pp. 383–411.
65. P. Taylor, M. Street, L. Wt, N. Carolina, Access details: Access Details: [subscription number 731847469] Communications in Soil Science and Plant Analysis Mehlich 3 soil test extractant: A modification of Mehlich 2 extractant, 37–41 (1984).
66. J. Major, M. Rondon, D. Molina, S. J. Riha, J. Lehmann, Nutrient Leaching in a Colombian savanna Oxisol amended with biochar. *J. Environ. Qual.* **41**, 1076–1086 (2012).
67. J. Balesdent, A. Mariotti, Measurement of soil organic matter turnover using  $^{13}\text{C}$  natural abundance, in *Mass Spectrometry Soils*, T. W. Boutton, S. Yamasaki, Eds. (Marcel Dekker, 1996), pp. 83–111.

**Acknowledgments:** We are grateful to the following persons of Hacienda La Cabaña S.A. for granting us access to field work, information, and hospitality: J. Zambrano, Y. Mejía, W. Mican, C. Colmenares, and others. We are also thankful to R. Schlaepfer and P. Mariotte from ECOS laboratory for the support in statistical analysis. **Funding:** This work was financed by the Swiss National Science Foundation (r4d-Ecosystems) "Oil Palm Adaptive Landscape" no. 152019. **Author contributions:** T.G., J.C.Q., and A.B. conceived and designed the study.

A.E. contributed to field work. J.C.Q. carried out the research assisted by T.G. J.C.Q. performed the statistical analysis with contributions from T.G. and A.B. J.C.Q. prepared the manuscript with contributions from all authors. T.G., A.B., A.E., and J.G. revised the manuscript. **Competing interests:** All authors declare that they have no competing interests. **Data and materials availability:** All data needed to evaluate the conclusions in the paper are present in the paper and/or the Supplementary Materials. Additional data related to this paper may be requested from the authors.

Submitted 20 December 2018  
Accepted 18 September 2019  
Published 20 November 2019  
10.1126/sciadv.aaw4418

**Citation:** J. C. Quezada, A. Etter, J. Ghazoul, A. Buttler, T. Guillaume, Carbon neutral expansion of oil palm plantations in the Neotropics. *Sci. Adv.* **5**, eaaw4418 (2019).

## Carbon neutral expansion of oil palm plantations in the Neotropics

Juan Carlos Quezada, Andres Etter, Jaboury Ghazoul, Alexandre Buttler and Thomas Guillaume

*Sci Adv* 5 (11), eaaw4418.  
DOI: 10.1126/sciadv.aaw4418

### ARTICLE TOOLS

<http://advances.sciencemag.org/content/5/11/eaaw4418>

### SUPPLEMENTARY MATERIALS

<http://advances.sciencemag.org/content/suppl/2019/11/18/5.11.eaaw4418.DC1>

### REFERENCES

This article cites 54 articles, 3 of which you can access for free  
<http://advances.sciencemag.org/content/5/11/eaaw4418#BIBL>

### PERMISSIONS

<http://www.sciencemag.org/help/reprints-and-permissions>

Use of this article is subject to the [Terms of Service](#)

---

*Science Advances* (ISSN 2375-2548) is published by the American Association for the Advancement of Science, 1200 New York Avenue NW, Washington, DC 20005. The title *Science Advances* is a registered trademark of AAAS.

Copyright © 2019 The Authors, some rights reserved; exclusive licensee American Association for the Advancement of Science. No claim to original U.S. Government Works. Distributed under a Creative Commons Attribution NonCommercial License 4.0 (CC BY-NC).