Short- and long-term carbon emissions from oil palm plantations converted from logged tropical peat swamp forest

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Abstract

Need for regional economic development and global demand for agro-industrial commodities has resulted in large-scale conversion of forested landscapes to industrial agriculture across South East Asia. However, net emissions of CO₂ from tropical peatland conversions may be significant and remain poorly quantified, resulting in controversy around the magnitude of carbon release following conversion. Here we present long term, whole ecosystem monitoring of carbon exchange from two oil palm plantations on converted tropical peat swamp forest. Our sites compare a newly converted oil palm plantation (OPnew) to a mature oil palm plantation.
(OPmature) and combine them in the context of existing emission factors. Mean annual net emission (NEE) of CO$_2$ measured at OPnew during the conversion period (137.8 Mg CO$_2$ ha$^{-1}$ yr$^{-1}$) were an order of magnitude lower during the measurement period at OPmature (17.5 Mg CO$_2$ ha$^{-1}$ yr$^{-1}$). However, mean water table depth (WTD) was shallower (0.26 m) than a typical drainage target of 0.6 m suggesting our emissions may be a conservative estimate for mature plantations, mean WTD at OPnew was more typical at 0.54 m. Reductions in net emissions were primarily driven by increasing biomass accumulation into highly productive palms. Further analysis suggested annual peat carbon losses of 24.9 Mg CO$_2$-C ha$^{-1}$ yr$^{-1}$ over the first 6 years, lower than previous estimates for this early period from subsidence studies, losses reduced to 12.8 Mg CO$_2$-C ha$^{-1}$ yr$^{-1}$ in the later, mature phase. Despite reductions in NEE and carbon loss over time, the system remained a large net source of carbon to the atmosphere after 12 years with the remaining 8 years of a typical plantation’s rotation unlikely to recoup losses. These results emphasise the need for effective protection of tropical peatlands globally and strengthening of legislative enforcement where moratoria on peatland conversion already exist.

**Introduction**

The need for economic development across South East Asia, and global demand for agro-industrial commodities such as palm oil, rubber and pulp wood, have driven the expansion of industrial scale agriculture and associated land-use change in recent decades. Agricultural crop production now covers 122 million hectares in the region (Kenney-Lazar and Ishikawa, 2019), around a quarter of the total land area.

In Malaysia and Indonesia alone, oil palm plantations now cover an estimated 23 million hectares (Miettinen et al., 2017, Gaveau et al., 2018) (Cheng et al., 2018). A significant proportion of this conversion has occurred recently on tropical peatlands; between 1990 and 2015, some 7.8 million hectares of these wetland peat swamp forests (PSF) were converted through forest clearance and land drainage (Miettinen et al., 2016). The economic contribution of expanding oil palm production, particularly in rural areas (Qaim et al., 2020), has come at a, yet to be fully determined, cost to the local environment and the global carbon balance.

Conversion of tropical forests, and particularly peat swamp forests, results in carbon emission (Couwenberg et al., 2009, Miettinen et al., 2017, Cook et al., 2018, Wijedasa et al.,
biomass loss (Kho and Jepsen, 2015), changes in carbon cycling dynamics (Swails et al., 2017) and the disturbance of previously stable soil carbon pools (Cheng, 2009, Kuzyakov, 2010). The majority of studies to date have employed subsidence and/or soil surface respiration measurements to estimate soil organic carbon (SOC) losses and estimates vary greatly, ranging (as CO$_2$-equivalent) from 20 to 100 Mg CO$_2$ ha$^{-1}$ yr$^{-1}$ (e.g. Wösten et al., 1997, Melling et al., 2005, Hooijer et al., 2012, Couwenberg and Hooijer, 2013, Dariah et al., 2014, Hergoualc’h et al., 2017, Ishikura et al., 2018, Manning et al., 2019, Cooper et al., 2020). While different experimental techniques and/or sampling designs may be influencing this variability, site-specific factors are thought to strongly contribute to the wide range of observed SOC losses.

Measurements from soil surface chambers and subsidence focus on SOC losses which is often advantageous, but these measurements do not allow direct monitoring of the net CO$_2$ change in the atmospheric carbon pool as they cannot concurrently capture photosynthetic carbon uptake and above ground sources of CO$_2$. In this regard, the eddy covariance (EC) technique (Baldocchi, 2003) provides distinct advantages: EC measures the net ecosystem exchange (NEE) of carbon, capturing both emission and uptake, and spatially integrates over complex intra-site sources, such as drainage ditch peat extraction and autotrophic respiration from plant biomass. EC has previously been employed in peatland forests in the region; Hirano et al. (2012) used it to investigate the impact of large scale anthropogenic disturbance on the carbon balance of tropical PSF in Indonesian Borneo, concluding that PSF were all now likely to be sources of atmospheric carbon (in the range of 7 to 18 Mg CO$_2$ ha$^{-1}$ yr$^{-1}$). A conclusion supported by another, very recent, EC study in logged PSF that showed a mean net emission of CO$_2$ over four years at 15.4 Mg CO$_2$ ha$^{-1}$ yr$^{-1}$ (Tang et al., 2020). However, the logistical and financial costs associated with EC have, to date, limited its deployment (Hill et al., 2017) and more studies are needed in tropical peatlands.

Only very recently have studies using EC started to report net carbon flux from oil palm plantations (Meijide et al., 2020), with only one monitoring oil palm cultivation on tropical peat (Kiew et al., 2020). The Kiew et al. (2020) study monitored a mature oil palm plantation on peatland in Sarawak, Malaysia and reported a mean annual net emission of 36.4 Mg CO$_2$ ha$^{-1}$ yr$^{-1}$, three times the Meijide et al. (2020) estimate for oil palm on mineral soils and double the emissions seen in even the most disturbed peat swamp forest reported in Hirano et al. (2012). Kiew et al. (2020) echoed Meijide et al. (2020) in calling for more eddy covariance studies on peatland plantations, particularly in the early years of conversion where net emissions are expected to be at their highest but are, as yet, unreported. The implications of this lack of EC monitoring of
different age classes of peatland oil palm is significant; Meijide et al. (2020) state they were unable to perform a full carbon Life Cycle Assessment (LCA) as a result, despite the need for better quantification being highlighted in earlier LCA studies of palm oil production (Mattsson et al., 2000, Schmidt, 2015).

In the absence of field studies of oil palm peatland conversions across the entire cultivation lifetime, emission factors determined for tropical forest conversion to agriculture on peatland have so far had to rely on very limited data. In deriving their Tier 1 emission factor of 40 Mg CO$_2$ ha$^{-1}$ yr$^{-1}$ for conversions to oil palm on drained peatland, the IPCC list only 8 direct studies, of which 6 were soil flux chamber studies and two were based on subsidence measurements. No ecosystem level monitoring of carbon flux was available to be included in the assessment. The IPCC (Hiraishi et al., 2014) stated that emissions during the early years of plantation establishment are expected to be significantly higher than their emission factor but were not included due to this lack of available data.

The absence of directly measured net carbon flux from peatland conversions to industrial plantation led to controversy around the GWP impacts of conversion (Wijedasa et al., 2017). Reviewers for an Environmental Protection Agency (EPA) report into peatland emission factors for oil palm (EPA 2014) were split over the importance of these early year emissions; debating the evidence of Hooijer et al. (2012) who had reported that very rapid subsidence recorded in the first five years of conversion was the result of large CO$_2$ emissions. There was a suggestion that compaction may be contributing more to this large initial subsidence than the Hooijer et al. (2012) study might suggest, and that there was not enough scientific evidence to the contrary.

Despite their limitations, current emission factors continue to play a crucial role in informing national and international policy. The industry standard Round Table on Sustainable Palm Oil (RSPO), the world’s largest certification initiative for palm oil (Qaim et al., 2020), rely on two synthesis studies, Hooijer et al. (2010) and Hooijer et al. (2012), as the key components in their assessment of the peat carbon impact of peatland conversion. The figure of CO$_2$ (Mg CO$_2$ ha$^{-1}$ yr$^{-1}$) emissions being 91 times water table depth (WTD) (m) from Hooijer et al. (2010) is the default value in their current GHG calculator (https://rspo.org/certification/palmghg/palm-ghg-calculator) which underpins estimates of peat decomposition in their certification scheme (https://rspo.org/certification). This coefficient of 91 was derived from a linear fit (with a fixed intercept of zero) to just eight data points collated from five studies, yet plays an important role in
modelling the global carbon budget (Houghton and Nassikas, 2017, Le Quéré et al., 2018, Friedlingstein et al., 2019).

In this study, we begin to address this important knowledge gap by contributing data collected by Eddy Covariance at two adjacent oil palm plantations established on tropical peatlands in South East Asia. One site captures a period following the initial conversion from peat swamp forest and another captures the mature phase. We present annual net ecosystem CO$_2$ fluxes from individual measurement years at both sites and partition them into photosynthetic uptake and whole ecosystem respiration. We then combine both datasets into a single chronosequence over a 151-month period and use a mass balance approach (incorporating estimates of biomass accumulation and forest residue decomposition) to calculate changes in soil carbon stocks. We present emission factors both for individual years and across relevant time periods (e.g. years 1-6, as highlighted by the IPCC). Finally, we investigate the relationship between soil water drainage and carbon loss in the context of previous emission coefficients and consider the potential impact of changes in plantation drainage targets.

**Methods**

**Site location and description**

The two study sites were individual blocks of commercially managed oil palm plantation situated within the Sabaju (OPnew: 3° 9.615’N, 113° 25.163’E) and Sebungan (OPmature: 3° 9.965’N, 113° 21.198’E) plantation estates in Sarawak, northern Malaysian Borneo. Climate is tropical equatorial with stable air temperatures (mean 26 °C) and high humidity and rainfall, typically ~3000 mm yr$^{-1}$. The sites are located 7.3 km from each other and represent typical oil palm plantation established on deep peat (up to 8 m) in the region. Both sites were established into previously logged peat swamp forest cleared and drained by cutting a regular network of drainage channels prior to palm establishment (see Cook et al. (2018) for a more detailed description of the plantation estates). The previously degraded forest at OPnew was cleared (without burning) at the beginning of 2016 with forest biomass cut and compacted on site and drainage channels cut into the peat. Establishment of oil palm (~160 plants ha$^{-1}$) was completed by the end of April 2016 and followed commercial practice throughout, no harvest was taken from the immature palms at OPnew during the study period. OPmature was established in July 2007 with fruit bunch
harvesting (FFB) beginning from month 32. EC monitoring at OPnew begins four months after forest clearance and continues for 41 months while at OPmature it begins 10 years after conversion and continues for 33 months.

**Instrumentation**

Eddy Covariance was carried out at both sites using identical instrumentation with the only significant difference being that profile measurements, for canopy storage of CO₂ and energy, were from three heights on a 20 m tower for the taller palms at OPmature compared to two on a 6 m tower at OPnew. LI-COR closed path systems (LI-7200/7550, *LI-COR Environmental, Lincoln, Nebraska, USA*) coupled to R3-50 Sonic Anemometer, *Gill Instruments Ltd, Lymington, Hampshire, UK*), were used at both sites, with sensors sited at the top of each tower. For OPmature this resulted in a measurement height (above ground) of 20.19 m, approximately 12 m above an 8 m canopy; for OPnew, sensors were at 6.06 m above a canopy that reached 2.6 m by the end of the study period. Prior to canopy development at OPnew topography was dominated by forest destruction residues compacted into rows of approximately 2 m in height which gave a typical measurement height above canopy of around 4 m. Canopy profile CO₂ and energy storage was measured using CO₂ diffusion sensors coupled with air and relative humidity sensors (*GMP343 and HMP155A, Vaisala Corporation, Helsinki, Finland*). For OPmature these were placed at 1 m, 6 m and 18 m above ground, for OPnew this was at 1 m and 6 m. Energy balance was monitored at two locations for each site using heat flux plates (*HFP01SC, Hukseflux Thermal Sensors, Delft, The Netherlands*) at 0.08 m soil depth coupled to soil moisture/temperature sensors (*Steven’s Hydraprobe, Stevens Water Monitoring Inc. Portland, Oregon, USA*) at 0.04 m. Water table depth (WTD) was monitored within 0.05 m diameter porous plastic pipe inserted to a depth of 2.5 m (*PX709GW submersible pressure transducer, Omega Engineering Inc. Norwalk, Connecticut, USA*). Precipitation was measured at the top of each tower using a tipping bucket gauge (*TR-525M, Texas Electronics, Dallas, Texas, USA*). EC data were collected at 10 Hz and stored to an industrial grade USB drive within the LI-7550, meteorological data at 1-minute intervals stored to Xlite 9210 dataloggers (*Satron Corporation, Sterling, Virginia, USA*).
Eddy Covariance data processing

Flux calculations

Raw flux data (10 Hz) are initially processed into 30-minute average CO$_2$ flux rates ($\mu$mol CO$_2$ m$^{-2}$ s$^{-1}$) using EddyPro software (v6.2.2 LI-COR Environmental, Lincoln, Nebraska, USA) before being storage corrected, gap-filled and further summed into mass integrations of NEE over time (e.g. Mg CO$_2$-C ha$^{-1}$ month$^{-1}$). Data handling, quality control and analyses was carried out using R (v3.5.1 R Core Team (2018), R Foundation for Statistical Computing, Vienna, Austria).

Statistical outliers (spikes) in the 10 Hz data were detected following Vickers and Mahrt (1997); vertical wind speed measurements were only accepted at $<5$ standard deviations ($\sigma$) from the 30-minute mean, other variables at $3.5\sigma$; 30-minute periods containing spikes at greater than 1% were flagged as poor quality. Time lags, discrepancies between precise sampling times at the anemometer and gas analyser, are compensated for using site specific covariance maximisation derived from data collected at the site. Detrending of turbulence fluctuations over each 30 minutes was through block averaging. Co-ordinate rotation, to accommodate imperfect alignment to the horizontal wind vector, was carried out through the planar fit method of Wilczak et al. (2001) using site derived parameters. Co-spectral analysis and correction of low- and high-pass filtering effects were carried out following Moncrieff et al. (1997) and Moncrieff et al. (2004). Spatial estimation of the areal source of sensor data capture (footprint assessment) followed Kljun et al. (2004) or Kormann and Meixner (2001) where turbulent friction was $<0.2$ m s$^{-1}$. CO$_2$ storage below the sensor height, which is not captured in turbulent eddy transfer through the EC sensor pathways, was accounted for through profile monitoring of CO$_2$ concentrations between the ground and sensor heights as outlined in Baldocchi et al. (2001): time stamped changes in absolute CO$_2$ concentration are captured by profile sensors and converted to volumetric ratios using the ideal gas law. These are then added to the corresponding flux measurements captured by EC at the half hour timestep.

Quality control flagging

Initial quality control flagging of each 30-minute flux average (statistical testing of the 10 Hz data) followed the Carbo-Europe standard 0-1-2 system of Foken et al. (2004). Zero being the
highest quality, values flagged at 2 were automatically discounted from further processing. Data spikes in the half-hourly processed CO₂ data were identified and removed following Papale et al. (2006) using the suggested median deviation threshold (z value) of 4. Absolute thresholds for sensible heat (H) were set between -200 and 350 W m⁻² and for latent energy (LE) at -50 to 500 W m⁻².

**Study site area (measurement fetch and footprint)**

The available study area which satisfied eddy covariance assumptions of homogeneity and representation of the area of interest (fetch) covered 41.7 ha at OPnew and 907 ha at OPmature. A combination of Google Earth (GE v7.3.2.5776, Imagery date 24/03/2016) and ARCGIS (ArcMap 10.5.1, ESRI, Redlands, CA, USA) was used in conjunction with the output from the footprint model to filter out any measurement periods where data collection extended beyond the ideal fetch. Taking the sensor tower location as a datum point, distances to the edge of the fetch boundary were measured at 10-degree increments. Half hourly output from the footprint model (percentage data contribution to total, distance to peak contribution and wind vector) were compared to these boundaries within 10-degree bins (total of 36) and considered acceptable where 70% of the information collected in each half-hour was sourced within the fetch boundary.

**Energy balance**

Energy balance closure (EBC) was investigated using an ordinary least square (OLS) regression at the half hour timestep between turbulent heat flux (latent energy (LE) plus sensible heat (H)) and available energy (net radiation plus soil heat flux). EBC was considered as the slope of the resulting OLS fit. Energy storage in relevant pools (canopy air space and soil volume) was calculated using specific heat capacity and moisture fluctuations. Energy lost to photosynthetic utilisation was calculated following Masseroni et al. (2014). The ratio between turbulent heat flux and available energy over the entire study period is presented as the Energy Balance Ratio (EBR) following Wilson et al. (2002).

**Gapfilling and flux partitioning**
Gapfilling of data rejected through quality control and partitioning of NEE into photosynthetic uptake (GPP) and ecosystem respiration ($R_{eco}$) was carried out using the ReddyProc package (Wutzler et al., 2018) within R. For gapfilling, this package utilises the mean diurnal separation (MDS) approach of Falge et al. (2001) with flux partitioning carried out using the light response curve method of Lasslop et al. (2010) to estimate daytime GPP, the sum of NEE and GPP being $R_{eco}$. Nighttime fluxes (below a global radiation ($R_g$) threshold of 20 W m$^{-2}$) are assumed solely $R_{eco}$ (Reichstein et al., 2005). Underestimation of fluxes during periods of insufficient turbulence was avoided by removing data recorded below site derived friction velocity thresholds ($u^*$ filtering) during the gapfilling process (Reichstein et al., 2005). Uncertainties in the half-hour fluxes are calculated, for gapfilled values, as the standard error of the mean of the values used to fill gaps. For retained original data these are artificially marked as gaps and again the standard error is calculated of the mean of values that would have been used to fill them. Standard errors are then propagated through cumulative sums.

**Chronosequence data series**

Data collection from OPnew starts from the beginning of September 2016 and runs to the end of January 2020, this represents months 5 to 45 in the plantation’s life cycle. For OPmature, data start from May 2017 and again run to January 2020, capturing months 119 to 151 of that plantation’s life cycle.

The first four months of data immediately following the conversion of PSF at OPnew were not collected due to the sensor installations not yet being in place. An estimation of these values has been made through modelling backwards at a monthly timestep from the first data available. For GPP an assumption is made that this would be zero immediately following conversion (dead forest residues and bare soil), therefore linear interpolation was carried out from a start point of zero at the beginning of May 2016 to the beginning of September 2016 (the first complete month’s data). For $R_{eco}$ a linear trend line was fitted to the existing $R_{eco}$ monthly dataset and extended back over these first four months.

Trends in fluxes (GPP, $R_{eco}$ and NEE) over the measurement period at each site are indicated by the slope of a linear model fitted against time, with significance accepted at $P<0.05$. Estimation of annual means within specific periods is calculated by multiplying the mean monthly
values for that period by twelve. A complete chronosequence of NEE, from months 1-151 was established using exponential interpolation (Stineman, 1980) between the OPnew and OPmature datasets at a monthly timestep. This interpolation was also applied to months 141 to 146 (April 2019 to August 2019) at OPmature where data were excluded due to sensor malfunction resulting in a gap too large for the MDS gapfilling routine.

Calculation of net primary productivity

Live standing biomass

Net primary productivity (NPP) is the sum of photosynthetic carbon sequestered into biomass pools on site during any given period. For live palm biomass carbon stocks, data were interpolated at a monthly timestep between biomass for age classes 3, 8 and 12 years, measured from destructive sampling from appropriately aged planting blocks at the Sabaju (age classes 3 and 8 years) and Sebungan plantations (age class 12 years) in 2019 and presented in Lewis et al. (2020). Individual palm component carbon stocks are summed to total palm biomass carbon using Eq.1. Root biomass was not directly sampled in Lewis et al. (2020) so has been assumed at 16% of total standing biomass following Khalid et al. (1999). This resulted in a timeseries of biomass carbon stocks across months 1-144; differences between values at the beginning and end of periods correlated to the EC flux measurements give NPP for that period (Eq.2).

Equation 1:

\[
\text{Carbon}_{\text{biomass}} = \text{Carbon}_{\text{roots}} + \text{Carbon}_{\text{trunks}} + \text{Carbon}_{\text{frodbases}} + \text{Carbon}_{\text{fronds}} + \text{Carbon}_{\text{FFB}} + \text{Carbon}_{\text{spear}} + \text{Carbon}_{\text{cabbage}} + \text{Carbon}_{\text{residual}}
\]

Where:

All components’ dry mass multiplied by fraction of carbon content (see Table S.1.2 in supplementary materials for full details)
Equation 2:

\[ NPP_{\text{period}} = \text{Carbon}_{\text{biomass}}(t) - \text{Carbon}_{\text{biomass}}(t-\text{period}) \]

Where:

\[ NPP = \text{net primary productivity (carbon sequestered into vegetative biomass, Mg CO}_2\text{-C ha}^{-1}\text{)} \]
\[ t = \text{time (period (months))} \]

Fresh Fruit Bunch harvest offtake

Harvest offtake, as fresh fruit bunch (FFB) was provided by the site managers for the OPmature planting block specifically within the Sebungan plantation at a monthly timestep from the date of first harvest in month 32 to month 144. For months 0-31 linear interpolation was used to complete the monthly timeseries from zero to first harvest. For sequestration calculations all FFB is considered to remain within the system (see Discussion), therefore total NPP of FFB for any given period is the cumulative sum of all harvest offtake during that period.

Pruned frond biomass

At each harvest, a number of fronds are cut to facilitate access to FFB and left in piles to decompose on site. While uptake of carbon into these fronds during growth, and return to atmosphere through decomposition, will be captured by EC in NEE, the carbon stored within the ecosystem in the total frond pile biomass at any given time needs to be accounted for in NPP. An estimate for this was derived from monthly interpolation between the numbers of frond bases (remnants of removed fronds) present on the palms at the 3-, 8- and 12-year time points. While multiplying these by a mean frond mass (for each age class of frond, similarly interpolated) gives an estimate of frond mass pruned in each month, account needs to be taken of the decomposition of each monthly addition to the pruned frond pile over the remaining study period. This was carried out by applying an exponential decay function, Eq.3, (Olson, 1963, Moradi et al., 2014) to
each monthly pruned frond mass and continuing to the end of the chronosequence, then summing across the remaining biomass from all previous prunings to a total pruned frond biomass pool per hectare for each month. The decomposition rate constant (fractional mass loss per month \(k\)) for frond biomass was set at 0.15, calculated from empirical measurements by Moradi et al. (2014).

**Equation 3:**

\[
mass_t = mass_{t-1} \exp(-kt)
\]

*Where:*

\(t\) = time (month)

\(k\) = decomposition rate constant (fractional mass loss per month)

**Calculation of forest debris decomposition**

The contribution to ecosystem respiration \(R_{eco}\) from the decomposition of the previous forest biomass \(R_f\) that was cut and compacted on-site prior to establishment of oil palm needs to be considered in the overall carbon budget as these emissions will be a significant contribution to the net flux captured by EC. Starting biomass and decomposition rate were not measured directly on site, so literature estimates have been used. As with the frond pile biomass, the decay rate for forest coarse woody debris (CWD) decomposition is calculated using Eq.3 at a monthly timestep, but in this case just considers a single biomass addition at the beginning of the conversion. Kho and Jepsen (2015) estimated 58.7 ± 10.7 Mg C ha\(^{-1}\) for logged peat swamp forest (as at OPnew); using their dry stem biomass carbon content of 47.4%, we derive forest biomass at clearance of 123.8 ± 22.6 Mg DM ha\(^{-1}\). Only one paper was found that monitored CWD decay under tropical peatland conditions in the same region, (Law et al., 2019), who were working in Sabah, Malaysian Borneo under very similar climatic conditions to this present study. That study reported a reported that after 12 months there had been a mean loss of 25.6% of the starting biomass. From this we calculated a rate constant \((k)\) at 0.02464 for decomposition of forest CWD at a monthly timestep which would result in the correct biomass loss by month 12.

**Changes in soil organic carbon**

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Changes in soil organic carbon ($\Delta$SOC) during specific periods were calculated as the difference between GPP and $R_{eco}$, taking account of sequestration of GPP into biomass stocks (NPP) and contribution of $R_{fr}$ to $R_{eco}$. Following production of a complete monthly timeseries of NPP and $R_{fr}$, as described above, periods corresponding to direct measurements of NEE (partitioned into GPP and $R_{eco}$) were used as parameters in Eq.4 to estimate peat carbon loss for the study period months at OPnew and OPmature. For annual carbon emission factors over the entire 151-month chronosequence of NEE, and for specified periods, Eq.4 is modified using Eq.5 (see Eq.S.2.1 in Supplementary Materials) and uses parameters derived by taking mean monthly values within selected time periods and multiplying by 12. Emission factors were calculated for years 0-6 (establishment), 6-12 (mature) and across the entire period. Additional exports of carbon ($\epsilon$), such as drainage losses of dissolved and particulate organic carbon or methane emissions ($CH_4$), are not captured in our eddy covariance results and are therefore not accounted for in these calculations, however, their potential magnitude is considered in the discussion section below.

Equation 4:

$$\Delta SOC = \frac{(GPP - (R_{eco} + NPP) + R_{fr}) - \epsilon}{t}$$

Where:

$\Delta SOC = \text{change in soil organic carbon (Mg CO}_2\text{C ha}^{-1})$

$GPP = \text{gross primary productivity (photosynthetic uptake of carbon, Mg CO}_2\text{C ha}^{-1})$

$R_{eco} = \text{Ecosystem respiration (Mg CO}_2\text{C ha}^{-1})$

$NPP = \text{net primary productivity (carbon sequestered into biomass, Mg CO}_2\text{C ha}^{-1})$

$R_{fr} = \text{respiration contribution from decomposition of forest residue (Mg CO}_2\text{C ha}^{-1})$

$\epsilon = \text{unaccounted factors (e.g. export as dissolved organic carbon, carbon content of emitted CH}_4\text{, etc., Mg CO}_2\text{C ha}^{-1})$

$t = \text{time (year)}$
Equation 5:

\[ \text{NEE} = R_{\text{eco}} - GPP \]

Where:

\( \text{NEE} = \text{net ecosystem exchange of carbon (Mg CO}_2\text{-C ha}^{-1} \) \)

**Relationship between peat carbon loss and water table depth**

The relationship between our estimate of SOC emission (as CO\(_2\)) and WTD was considered in two separate analyses. Firstly, at an annual timestep in the context of the Hooijer *et al.* (2010) analysis, by the recreation of their original linear regression (from data provided in Hooijer *et al.* (2006)) and subsequent inclusion of annual \(\Delta \text{SOC}\) from OPnew and OPmature. Secondly, taking advantage of our high frequency respiration data (at the half hour timestep) at OPnew, we fit a non-linear 2\(^{nd}\) order polynomial curve to nighttime NEE, assumed to be entirely respiration, and WTD following de-trending of the data series and binning of \(R_{\text{eco}}\) into 0.01 m increments of WTD. Only original measured data (not gapfilled) were used and selected at the highest quality (qc flagged at zero (Mauder and Foken, 2004)). The output of the model fit was then used to predict respiration rates for 0.1 m WTD increments between 0 and 0.8 m below the soil surface (the measured WTD range at OPnew within the study period).

**Results**

*See supplementary materials for full details of eddy covariance data capture and retention following quality control. Values ± given throughout these results indicate the standard error of the mean (S.E.M.)*

**Climate data**
As expected in equatorial, tropical climates rainfall was high and temperatures were relatively stable over time with only a minimal seasonal component (Fig. 1). Rainfall averaged 2856 ± 96 mm yr\(^{-1}\) across the two sites, and had a mean temperature of 26.9 ± 0.03 °C. The exposed soils at OPnew were on average 2 °C warmer than at OPmature, with a mean soil temperature in the upper 0.04 m of 30.3 ± 0.03 °C compared with 28.3 ± 0.03 °C under the canopy at OPmature. The mean water table depth was 0.26 ± 0.04 m at OPmature and 0.54 ± 0.05 m at OPnew.

**Measured carbon flux (as CO\(_2\))**

**Ecosystem respiration**

A small, but statistically significant, difference was seen between mean monthly \(R_{eco}\) at the two sites (Welch Two Sample T-test, \(t = 7.2, df = 47.4, p < 0.0001\)). OPnew was higher at 17.6 ± 0.4 Mg CO\(_2\) ha\(^{-1}\) mth\(^{-1}\) (± standard error of the mean of monthly totals) compared to OPmature 14.2 ± 0.4 Mg CO\(_2\) ha\(^{-1}\) mth\(^{-1}\). For OPnew there was no significant change in monthly \(R_{eco}\) over the monitoring period (\(F = 0.6, df = 39, p = 0.4\)); in contrast, at OPmature there was a slight, but significant, decline in \(R_{eco}\) over time, reducing by 0.11 Mg CO\(_2\) ha\(^{-1}\) mth\(^{-1}\) (\(f = 6.35, df = 25, p = 0.02\)).

**Photosynthetic uptake**

Monthly GPP at OPnew showed a significant increase over the study period (\(f = 194.7, df = 39, p < 0.0001\)); from 1.1 Mg CO\(_2\) ha\(^{-1}\) mth\(^{-1}\) in September 2016 (four months after conversion) it increased by 0.18 Mg CO\(_2\) ha\(^{-1}\) mth\(^{-1}\). GPP at OPmature also changed over time, but in this case showed a small, but significant, reduction in uptake; declining over the study period by 0.07 Mg CO\(_2\) ha\(^{-1}\) mth\(^{-1}\) (\(f = 6.35, df = 25, p < 0.05\)). Mean monthly GPP was 6.2 ± 0.4 Mg CO\(_2\) ha\(^{-1}\) mth\(^{-1}\) at OPnew and 12.8 ± 0.2 Mg CO\(_2\) ha\(^{-1}\) mth\(^{-1}\) at OPmature (Fig.2).

**Net ecosystem exchange**

Both sites were cumulative net sources of carbon to the atmosphere, with a mean annual NEE calculated over the entire study period of 137.8 ± 4.9 Mg CO\(_2\) ha\(^{-1}\) yr\(^{-1}\) at OPnew and 17.5 ± 2.1.
Mg CO$_2$ ha$^{-1}$ yr$^{-1}$ at OPmature. Only two monthly totals of NEE showed a net uptake, and both were at OPmature: November 2018 at -0.3 Mg CO$_2$ ha$^{-1}$ mth$^{-1}$ and September 2019 at -0.6 Mg CO$_2$ ha$^{-1}$ mth$^{-1}$. NEE decreased significantly during the study period at OPnew ($f=98.9$, $df=39$, $p<0.0001$), reducing by 18.3 Mg ha$^{-1}$ CO$_2$ mth$^{-1}$; in contrast, due to corresponding decreases in both GPP and $R_{eco}$, NEE did not change significantly over time at OPmature ($f=2.2$, $df=25$, $p=0.16$). Table 1 shows NEE for individual 12-month periods captured at both study sites.

Soil organic carbon loss (as CO$_2$-C) for individual study years at both sites

The change in soil organic carbon ($\Delta$SOC) was 2.5 to 3 times higher in OPnew compared to OPmature (Table 1). The sequestration rate of carbon into the extant biomass pool (NPP), increased between years 1 and 3 of the study at OPnew from 0.8 ± 0.07 Mg CO$_2$-C ha$^{-1}$ yr$^{-1}$ (plantation cycle months 5-16) to 2.3 ± 0.3 Mg CO$_2$-C ha$^{-1}$ yr$^{-1}$ (months 29-40). At OPmature NPP declined slightly over the two years of monitoring, from 7.2 ± 2.7 Mg CO$_2$-C ha$^{-1}$ yr$^{-1}$ (plantation cycle months 119-130) to 6.8 ± 3.0 Mg CO$_2$-C ha$^{-1}$ yr$^{-1}$ (months 131-142). The estimated contribution of respiration from the decomposition of forest biomass ($R_{fr}$) to gross ecosystem respiration ($R_{eco}$) over the study periods at each site reduced from a mean of 19 % at OPnew to 1.5 % at OPmature.

Chronosequence of cumulative NEE

Total cumulative NEE across the entire chronosequence suggested a net emission of CO$_2$ from the site at 823.3 ± 0.9 Mg CO$_2$ ha$^{-1}$ (224.9 ± 0.2 Mg CO$_2$-C ha$^{-1}$) after 151 months. As can be seen in Fig.3, interpolating across the missing months in the OPmature dataset (months 141 to 146) suggested the system might have been showing a net monthly uptake during this period which would offset the rate of increase in NEE. This can be seen in the corresponding levelling off in the cumulative NEE curve beyond month 140. The first four months following conversion, extrapolated backwards from the beginning of monitoring at OPnew, month 5 (see methods), suggested total $R_{eco}$ during that period of 68.8 Mg CO$_2$ ha$^{-1}$ with GPP during the same four month period at 1.6 Mg CO$_2$ ha$^{-1}$; resulting in an estimated net emission (NEE) of 67.2 ± 0.5 Mg CO$_2$ ha$^{-1}$ over this first four months. Adding these months (1 to 4) to the beginning of the monthly time
series for OPnew (months 5 to 45) gave a net cumulative NEE by the end of the OPnew monitoring period at 536.7 ± 0.6 Mg CO\textsubscript{2} ha\textsuperscript{-1} (146.7 ± 0.2 Mg CO\textsubscript{2}-C ha\textsuperscript{-1}).

Table 1. Annual carbon fluxes and biomass uptake (Mg CO\textsubscript{2}-C ha\textsuperscript{-1} yr\textsuperscript{-1}) for complete measurement years during the study period and resulting estimates of changes in soil organic carbon (ΔSOC) calculated using Eq.4. (ΔSOC = change in soil organic carbon during period, R\textsubscript{fr} = respiration of carbon from decomposition of forest residue, GPP = gross primary productivity, R\textsubscript{eco} = total ecosystem respiration, NEE = net ecosystem exchange of carbon, NPP = net primary productivity).
### Plantation cycle months

| Plantation cycle months | 5-16 | 17-28 | 29-40 | 119-130 | 131-142 |
|-------------------------|------|-------|-------|---------|---------|
| \( \Delta \text{SOC} \) | -33.4 | -28.3 | -29.2 | -11.3 | -12.2 |
| \( \pm 7.5 \) | \( \pm 1.3 \) | \( \pm 1.8 \) | \( \pm 2.7 \) | \( \pm 3.0 \) |
| \( R_{fr} \) | 13.9 | 10.4 | 7.7 | 0.8 | 0.6 |
| \( \pm 0.7 \) | \( \pm 0.6 \) | \( \pm 0.4 \) | \( \pm 0.04 \) | \( \pm 0.03 \) |
| \( GPP \) | 9.9 | 19.7 | 26.2 | 44.1 | 40.3 |
| \( \pm 7.4 \) | \( \pm 1.1 \) | \( \pm 1.8 \) | \( \pm 0.02 \) | \( \pm 0.05 \) |
| \( R_{eco} \) | 56.4 | 57.4 | 60.9 | 49.00 | 46.3 |
| \( \pm 0.03 \) | \( \pm 0.05 \) | \( \pm 0.05 \) | \( \pm 0.02 \) | \( \pm 0.02 \) |
| \( \text{NEE} \) | 46.5 | 37.7 | 34.7 | 4.9 | 6.0 |
| \( \pm 0.03 \) | \( \pm 0.03 \) | \( \pm 0.06 \) | \( \pm 0.05 \) | \( \pm 0.06 \) |
| \( \text{NPP} \) | 0.8 | 1.0 | 2.3 | 7.2 | 6.8 |
| \( \pm 0.07 \) | \( \pm 0.1 \) | \( \pm 0.3 \) | \( \pm 2.7 \) | \( \pm 3.0 \) |

Mass units: Mg CO\(_2\)-C ha\(^{-1}\) yr\(^{-1}\) (\(\pm\) propagated S.E.)

### Annual carbon emission factors (as CO\(_2\)-C)

Mean annual NEE across the 151-month chronosequence showed a net annual emission of 17.9 ± 1.3 Mg CO\(_2\)-C ha\(^{-1}\) yr\(^{-1}\) (65.6 ± 4.8 Mg CO\(_2\) ha\(^{-1}\) yr\(^{-1}\)). Using Eq.4 (modified by Eq.5) to account for carbon sequestered into on-site biomass at a mean annual NPP of 4.9 ± 0.2 Mg CO\(_2\)-C ha\(^{-1}\) yr\(^{-1}\) and a mean annual carbon emission from the decomposition of forest residue (\(R_{fr}\)) at 4.6 ± 0.1 Mg CO\(_2\)-C ha\(^{-1}\), suggested a mean \(\Delta \text{SOC}\) over the entire period at -18.3 ± 1.3 Mg CO\(_2\)-C ha\(^{-1}\) yr\(^{-1}\) (equivalent to a soil surface emission of 67 ± 4.8 Mg CO\(_2\) ha\(^{-1}\) yr\(^{-1}\)).

For years 1-6 (months 1-72), mean annual NEE was 30.2 ± 0.1 Mg CO\(_2\)-C ha\(^{-1}\) yr\(^{-1}\), mean annual NPP was 2.9 ± 1.3 Mg CO\(_2\)-C ha\(^{-1}\) yr\(^{-1}\) and \(R_{fr}\) was 8.1 ± 0.2 Mg CO\(_2\)-C ha\(^{-1}\) yr\(^{-1}\) which
resulted in an early years emission factor for peat carbon at \(-24.9 \pm 1.3 \text{ Mg CO}_2\text{-C ha}^{-1} \text{ yr}^{-1}\) \((91.4 \pm 4.8 \text{ Mg CO}_2\text{-C ha}^{-1} \text{ yr}^{-1})\).

This was much reduced for years 7 to 12 (months 73 to 144), with NEE at \(7.2 \pm 0.4 \text{ Mg CO}_2\text{-C ha}^{-1} \text{ yr}^{-1}\), NPP at \(6.9 \pm 6.1 \text{ Mg CO}_2\text{-C ha}^{-1} \text{ yr}^{-1}\) and \(R\) at \(1.4 \pm 0.01\) which resulted in a mature phase emission factor of \(-12.8 \pm 6.1 \text{ Mg CO}_2\text{-C ha}^{-1} \text{ yr}^{-1}\) \((46.9 \pm 22.3 \text{ Mg CO}_2\text{-C ha}^{-1} \text{ yr}^{-1})\).

Table 2 shows annual and cumulative components for the chronosequence, (NEE, NPP and \(R\)) with the resulting estimates of \(\Delta\text{SOC}\) for years 1 to 12 (mths 1 to 144).

**Linear fit of \(\Delta\text{SOC}\) (as CO\(_2\) flux) to water table depth**

Adding our sites’ annual soil carbon emissions (\(\Delta\text{SOC}\) in Table 1, as CO\(_2\)) and mean WTD (see Fig.4) to the Hooijer et al. (2010) dataset increased the model coefficient to \(\text{CO}_2 = 118.1 \pm 14.5 \times \text{WTD (m)}\), with a highly significant fit \((F=66.48, df=12, p<0.0001)\). Allowing the intercept to solve gave a model fit of \(\text{CO}_2 = 7.4 + 105.6 \times \text{WTD (m)}\) but only the slope was found to be significant \((p=0.04)\). Excluding the very high fluxes seen from the conversion period at OPnew, by including only fluxes from OPmature (see supplementary material Fig. S.1.5), resulted in a model fit in close agreement with the original Hooijer et al. (2010) analysis with a highly significant model fit at \(\text{CO}_2 = 93.7 \pm 9.8 \times \text{WTD (m)}\) \((F=91.4, df=9, p<0.0001)\).

**Non-linear fit of ecosystem respiration to water table depth**

The 2\(^{nd}\) order polynomial fit of nighttime \(R_{\text{eco}}\) to WTD (see Fig. S.1.6. in supplementary) was highly significant with an adjusted \(R^2\) of 0.83 \((F = 157.6, df = 64, P < 0.0001)\). Model coefficients across the range of WTD found at OPnew (0 – 0.8 m below the surface, binned into 0.1 m increments) predicted a 154% increase in annual \(R_{\text{eco}}\) when moving from the shallowest WTD (0 m) to the deepest (0.8 m). Table 3 shows predicted annual \(R_{\text{eco}}\) for each 0.1 m WTD increment and their magnitude relative to a typical plantation drainage target of 0.6 m. Fig. 5 shows a graphical representation of these estimated percentage changes in annual \(R_{\text{eco}}\) when comparing different mean annual water table depths. For example, raising WTD to 0.2 m would see a 31% reduction in respiration compared to the typical 0.6 m target. In contrast, differences between depth increments deeper than 0.6 m were insignificant.
Table 2. Chronosequence estimates of the change in soil organic carbon and individual components of the mass balance equation (Eq.4 modified by Eq.5) for individual years up to year 12 (months 1 to 144) and cumulatively over this period. (NEE = net ecosystem exchange of carbon, NPP = net primary productivity, \(R_f\) = forest residue decomposition, \(\Delta SOC\) = change in soil organic carbon over period, units are mass of carbon: Mg CO\(_2\)-C ha\(^{-1}\))

| Plantation cycle year | 1    | 2    | 3    | 4    | 5    | 6    | 7    | 8    | 9    | 10   | 11   | 12   |
|-----------------------|------|------|------|------|------|------|------|------|------|------|------|------|
| **NEE annual**        | 50.9 | 39.2 | 36.2 | 25.2 | 16.2 | 13.6 | 11.5 | 9.5  | 7.6  | 5.7  | 5.6  | 3.4  |
| ± 0.2                 | ± 0.03 | ± 0.04 | ± 0.08 | ± 0.08 | ± 0.06 | ± 0.05 | ± 0.04 | ± 0.04 | ± 0.03 | ± 0.05 | ± 0.05 |
| **NEE cumulative**    | 50.9 | 90.1 | 126.2 | 151.4 | 167.5 | 181.1 | 192.7 | 202.2 | 209.8 | 215.5 | 221.0 | 224.5 |
| ± 0.2                 | ± 0.2 | ± 0.2 | ± 0.2 | ± 0.2 | ± 0.2 | ± 0.2 | ± 0.2 | ± 0.2 | ± 0.2 | ± 0.2 | ± 0.2 | ± 0.2 |
| **NPP annual**        | 0.7  | 1.0  | 1.2  | 4.6  | 4.9  | 5.1  | 6.1  | 6.4  | 6.9  | 6.8  | 7.9  | 7.6  |
| ± 0.1                 | ± 0.3 | ± 0.5 | ± 1.2 | ± 2.3 | ± 3.4 | ± 4.5 | ± 5.6 | ± 6.0 | ± 6.3 | ± 6.8 | ± 7.3 |
| **NPP cumulative**    | 0.7  | 1.7  | 2.9  | 7.5  | 12.4 | 17.4 | 23.5 | 29.9 | 36.8 | 43.6 | 51.5 | 59.1 |
| ± 0.1                 | ± 0.3 | ± 0.6 | ± 1.3 | ± 2.7 | ± 4.3 | ± 6.2 | ± 8.3 | ± 10.3 | ± 12.1 | ± 13.8 | ± 15.7 |
| **\(R_f\) annual**   | 15.0 | 11.2 | 8.3  | 6.2  | 4.6  | 3.4  | 2.6  | 1.9  | 1.4  | 1.1  | 0.8  | 0.6  |
| ± 0.7                 | ± 0.4 | ± 0.2 | ± 0.1 | ± 0.06 | ± 0.04 | ± 0.02 | ± 0.01 | ± 0.01 | ± 0.003 | ± 0.002 | ± 0.001 |
| **\(R_f\) cumulative**| 15.0 | 26.2 | 34.5 | 40.7 | 45.3 | 48.8 | 51.3 | 53.2 | 54.6 | 55.6 | 56.4 | 57.0 |
| ± 0.7                 | ± 0.8 | ± 0.8 | ± 0.8 | ± 0.8 | ± 0.8 | ± 0.8 | ± 0.8 | ± 0.8 | ± 0.8 | ± 0.8 | ± 0.8 | ± 0.8 |
| **\(\Delta SOC\) annual**| -36.6 | -29.0 | -29.0 | -23.6 | -16.4 | -15.3 | -15.0 | -14.0 | -13.1 | -11.4 | -12.6 | -10.5 |
| ± 0.7                 | ± 0.5 | ± 0.5 | ± 1.2 | ± 2.3 | ± 3.4 | ± 4.5 | ± 5.6 | ± 6.0 | ± 6.3 | ± 6.8 | ± 7.3 |
| ASOC cumulative | -36.6 | -65.6 | -94.6 | -118.2 | -134.6± | -149.8 | -164.9 | -178.9 | -192.00 | -203.4 | -216.0 | -226.5 |
|-----------------|-------|-------|-------|--------|---------|---------|---------|---------|----------|---------|---------|---------|
|                 | ±0.7  | ±0.9  | ±1.0  | ±1.6   | ±2.8    | ±4.4    | ±6.3    | ±8.4    | ±10.3    | ±12.1   | ±13.9   | ±15.7   |

**Mass units:** Mg CO$_2$-C ha$^{-1}$ yr$^{-1}$ (± propagated S.E.)
Table 3. Modelled relationship between ecosystem respiration ($R_{eco}$) and water table depth (WTD)

| WTD [m] | Annual $R_{eco}$ (CO₂-C) [Mg ha⁻¹ yr⁻¹] | Flux relative to WTD 0.6m [%] |
|---------|----------------------------------------|-------------------------------|
| 0       | 22.2 ± 4.8                             | 40.4                          |
| 0.1     | 30.8 ± 2.9                             | 55.8                          |
| 0.2     | 38.0 ± 1.6                             | 69.1                          |
| 0.3     | 44.1 ± 1.0                             | 80.1                          |
| 0.4     | 49.0 ± 1.1                             | 89.0                          |
| 0.5     | 52.7 ± 1.1                             | 95.6                          |
| 0.6     | 55.1 ± 1.0                             | 100.0                         |
| 0.7     | 56.3 ± 1.2                             | 102.3                         |
| 0.8     | 56.4 ± 2.1                             | 102.3                         |

± indicates the 95% confidence interval of the model fit

Discussion

CO₂ flux

We have presented, for the first time, a comparative study of measured net ecosystem exchange of carbon (NEE) as CO₂ between the initial years of peatland conversion to oil palm and the later mature phase. Our results show the dramatic difference between the very high early conversion period annual emission of CO₂ to the atmosphere at 137.8 Mg CO₂ ha⁻¹ yr⁻¹, and the mature phase emission at 17.5 Mg CO₂ ha⁻¹ yr⁻¹. Our mature phase figure for NEE is reasonably
consistent with the estimated value for mature peatland of $12.1 \pm 10.2$ Mg CO$_2$ ha$^{-1}$ yr$^{-1}$ from Meijide et al. (2020), though less than half the $36.4$ Mg CO$_2$ ha$^{-1}$ yr$^{-1}$ measured by Kiew et al. (2020) at their mature peatland plantation. Our 12.5-year chronosequence demonstrated that while the much lower NEE in later years might level off the rate of increase in cumulative NEE, it was highly unlikely to offset that cumulative carbon emission over a plantation lifetime. Recouping the total emissions over our chronosequence period (around $800$ Mg CO$_2$ ha$^{-1}$) would require an average net uptake of $\sim 100$ Mg CO$_2$ ha$^{-1}$ yr$^{-1}$ for the remaining $7.5$ years of a typical $20$-year plantation lifetime.

While our results show that conversion may be adding around $110$ Mg CO$_2$ ha$^{-1}$ yr$^{-1}$ ($30.2$ Mg CO$_2$-C ha$^{-1}$ yr$^{-1}$) to the atmosphere in the first six years of the conversion (months 1-72), not all of this would be coming directly from soil carbon decomposition. Our simple forest decomposition model suggested that around a quarter of this net emission could have been coming from CO$_2$ released by the decay of forest biomass. In contrast, while NEE had dropped to around $26$ Mg CO$_2$ ha$^{-1}$ yr$^{-1}$ over the next six years (months 73-144), soil carbon emission remained high at around $47$ Mg CO$_2$ ha$^{-1}$ yr$^{-1}$ but was being masked by NPP. Sequestration of carbon into the biomass pool (NPP) was equivalent to around $54\%$ of the peat carbon loss for that period. A small proportion of that carbon will be contained within the fruit harvest offtake, removed from the site and returned to the atmosphere during oil production and consumption, the remainder will be held within the palm and frond litter biomass pool until re-cultivation (typically at around year 20) where it will begin to return to the atmosphere during the decomposition of the palm biomass following clearance.

We also considered the relative contributions to NEE from uptake (GPP) and emission ($R_{eco}$). The common approach for partitioning NEE (Reichstein et al., 2005) uses parameterisation of the relationship between nighttime NEE (assumed to be entirely $R_{eco}$) and air temperature to estimate the contribution of $R_{eco}$ to daytime NEE (the residual being GPP). However, this approach relies on a strong relationship between respiration and temperature. This can be problematic in tropical climates where temperature ranges (both diurnally and seasonally) tend to be very much narrower than in temperate zones, and likely compounded by the strong relationship in drained peatlands between $R_{eco}$ and WTD. As an alternative, we adopted the approach of Lasslop et al. (2010) using a light response curve fitted to daytime NEE to estimate GPP. Kiew et al. (2020) concluded that low GPP in their poorly established plantation was responsible for their
large on-site net emissions. In line with their conclusion, as seen in Fig.2, while $R_{eco}$ was slightly lower at our mature site compared to OPnew, it was the much higher GPP into the mature palms that was primarily responsible for driving this reduction in NEE. There was a small, but statistically significant, reduction in both GPP and $R_{eco}$ over time during the OPmature monitoring period, which appears to be most apparent in the period between August 2018 and February 2019. This dip in activity appears to have recovered at some point during our missing data period between then and September 2019. Stiegler et al. (2019) showed that drought conditions resulting from an El Nino-Southern Oscillation (ENSO) event in 2015 led to reduced CO$_2$ uptake into their study site at an Indonesian oil palm plantation. This might suggest that another ENSO event in the region, recorded between September 2018 and June 2019 (https://www.metoffice.gov.uk/research/climate/seasonal-to-decadal/gpc-outlooks/el-nino-la-nina), could be linked to our indication of a drop off in photosynthetic activity. The later period of this ENSO event also coincided with a particularly bad period of air pollution haze due to extensive vegetation burning across the entire region (https://www.bbc.co.uk/news/world-asia-34265922), which may also have contributed to this. Monthly yield data from the site (not published) suggest a corresponding dip in FFB harvest during this period which might corroborate this, though more detailed analysis would be required to reach any firm conclusion.

**Relationship between WTD and soil carbon loss (as CO$_2$)**

Adding our estimate of soil carbon loss (as CO$_2$) from OPmature to the Hooijer et al. (2010) linear regression model we found that our mature site fitted remarkably well within their original dataset, only raising the coefficient to 93.7 from their $CO_2 = 91 * WTD$. However, including our early conversion period at OPnew increased this sensitivity estimate by 26%. This reinforces the importance of incorporating the early years of conversion into assessments of the carbon impacts of peatland conversion and emphasises the need for emission factors covering entire cropping periods.

A single coefficient such as this may, though, be too simplistic. As discussed in Hooijer et al., (2006), a coefficient for the relationship between CO$_2$ flux and WTD is unlikely to be consistent throughout the soil profile, and it was a lack of available data which limited their
original analysis to a simple linear fit. In their discussion (see note under figure 12 in Hooijer et al. (2006)), the authors suggest that CO$_2$ emissions might be reduced at WTD of 0.2 - 0.3 m and at zero when WTD = 0, i.e. waterlogged conditions would promote the formation of peat and net CO$_2$ emissions would <= 0. Their suggestion that a linear coefficient of 91 would hold true for WTD from 0.25 and 1.1 m was discussed in Couwenberg et al. (2010)) who concluded there was not enough evidence to clearly state whether rates of subsidence (as a measure of peat decomposition) became static beyond a WTD of 0.5 m.

Taking advantage of our high frequency, long term dataset at OPnew, where heterotrophic respiration should heavily outweigh the limited autotrophic respiration from immature, widely spaced palms, we investigated this relationship using a non-linear, polynomial fit. Here we found not only a highly significant relationship (in contrast to Cooper et al. (2020), whose study had very little regression data available) but one that implied the opposite to what was suggested in Hooijer et al. (2006). We found a much greater sensitivity to drainage in the upper half of the soil profile compared to the lower half over our 0 - 0.8 m WTD range (see Table 3). Each additional 0.1 m drainage within the upper 0.4 m produced an additional 24.5 Mg CO$_2$ ha$^{-1}$ yr$^{-1}$ (compared to 9.1 Mg CO$_2$ ha$^{-1}$ yr$^{-1}$ expected from Hooijer et al. (2010)), while between 0.5 to 0.8 m this decreased to 6.8 Mg CO$_2$ ha$^{-1}$ yr$^{-1}$. This trend may be reasonably intuitive, Leifeld et al. (2012) showed that peat decomposition rates were dependant on organic matter quality and that decomposability was higher in the newer organic matter nearer the peat surface. Given the importance of temperature in driving soil carbon decomposition (Lloyd and Taylor, 1994), it would also follow that the drained upper layers (with a corresponding decrease in soil heat capacity) would see greater (and faster) soil temperature responses to incoming solar radiation, again suggesting that we might expect this greater sensitivity of respiration to drainage in the upper profile. While our results (Fig. 5) are only from a single site, they do indicate the potential impact, (and significant carbon conservation) that might be achieved through more strategic management of water table in the upper soil layers. However, more work is needed to investigate the impact that reducing WTD would have on fruit yield.

Comparison to subsidence studies

Hooijer et al. (2012) estimated (from subsidence studies) that the first five years of conversion from PSF to plantation (acacia as this is the conversion they used to estimate 0-5 yr
fluxes) would see a mean loss of peat carbon at 48.6 Mg CO$_2$-C ha$^{-1}$ yr$^{-1}$ (calculated from CO$_{2eq}$ figures in their table.2), our estimate for OPnew mean peat carbon loss was considerably lower than this at 30.3 Mg CO$_2$-C ha$^{-1}$ yr$^{-1}$ over the first three years. For mature oil palm sites (>6 yrs) they suggest a mean annual loss of 19.9 Mg CO$_2$-C ha$^{-1}$ yr$^{-1}$ at a mean WTD of 0.71 m (therefore CO$_{2eq}$ = 102.6 * WTD (m)), a figure in close agreement with a modelled range of 18 to 22 Mg CO$_2$-C ha$^{-1}$ yr$^{-1}$ at WTD of 0.7 m given in Carlson et al. (2015). The OPmature site in our study (years 11 and 12) again showed a lower carbon loss than this at only 11.7 Mg CO$_2$-C ha$^{-1}$ yr$^{-1}$. However, mean WTD at OPmature was much closer to the soil surface (0.26 m) than in the Hooijer et al. (2012) analysis, which gives a relationship of CO$_2$ = 164.7 * WTD (m), around 60% higher, which might be expected from our comparison between the polynomial and linear fits discussed above.

Hooijer et al. (2012) estimated a long term (yrs 0-18) mean annual carbon loss of 32.5 Mg CO$_2$-C ha$^{-1}$ yr$^{-1}$, a figure that agrees well with an annualised 20 year figure of 29 Mg CO$_2$-C ha$^{-1}$ yr$^{-1}$ calculated through literature review by Page et al. (2011). These values are both heavily influenced by the inclusion of very high peat carbon emissions in the early years estimated from observed rapid initial subsidence. Our chronosequence estimate of mean annual soil carbon loss (over 12 years) was 35% lower than this at 18.9 Mg CO$_2$-C ha$^{-1}$ yr$^{-1}$, a figure in very close agreement to the mean annual carbon loss for conversions estimated in Couwenberg and Hooijer (2013) at 18 Mg CO$_2$-C ha$^{-1}$ yr$^{-1}$, a study aimed at improving the methodology for subsidence estimates of carbon loss from peatlands conversions. However, this later study assumed, but did not account for, peat carbon losses in the early years being far higher and the figure of 18 Mg ha$^{-1}$ yr$^{-1}$ was recommended only for mature conversions in a ‘steady state’. Our results for the early years, which are twice the mature value, are included in our 12 year estimate, suggesting, from our sites at least, that including these early years does not raise the mean emission beyond the Couwenberg and Hooijer (2013) estimate. However, we must acknowledge that drainage at our mature site particularly may not have been as effective as site managers would typically prefer. A recent meta-analysis (Prananto et al., 2020) showed a mean WTD across 138 tropical peatland plantations at 0.56 m, 0.3 m deeper than our mean for OPmature (0.26 m). With reference to Fig. 5, increasing WTD from 0.3 m to 0.6 m below the surface could see an increase in CO$_2$ emission of up to 25%, suggesting that emissions for OPmature may be lower than might be expected from deeper-drained plantations.
It should also be noted that our estimate of CO$_2$ flux from the decomposition of the forest biomass was contributing around a quarter of the total ecosystem CO$_2$ emission to the atmosphere ($R_{co}$). This contribution would be reducing the estimate of early years peat decomposition considerably in our mass balance equation, 75% of the forest residue was decomposed within 4 years in our decomposition model. This estimate was based on a literature figure for the starting biomass (Kho and Jepsen, 2015) and an assumption that 25% had decomposed by the end of the first year from a single decomposition study (Law et al., 2019). Our decomposition rate can be compared to a study carried out under similar climatic conditions in Panama, South America (Hoyos-Santillan et al., 2015), which reported 44% of starting biomass remaining after 2 years (stems up to 0.05 m diameter, when left above ground), our chosen decay constant would result in 55% remaining after the same period. Given that our assumption includes CWD over a range of diameters, including much larger than 0.05 m, this might seem a reasonable value, however it is an assumption. Any decrease in decomposition rate or starting biomass would have a corresponding increase in the estimate of SOC loss for these early years. Our sensitivity analysis (see Supplementary materials S.2) showed that while adjusting these values did have this impact on estimates of $\Delta$SOC, particularly in the early years, even at unrealistic levels they did not bring emissions from our sites to the levels suggested in subsidence studies. Uncertainty levels were also high in our estimate of NPP and increased over time with propagation of all the uncertainties in individual vegetation component assessments. Again, the implications of this can be seen in our sensitivity analysis in Table S.2.2 in Supplementary materials where we consider the impact of doubling our estimate of NPP across a range of residue decomposition rate scenarios. Even at our lowest decomposition rate and doubled NPP our long-term estimate of peat carbon loss remained 18% lower than the corresponding subsidence estimate.

An aspect not captured in our direct monitoring of the ecosystem/atmosphere exchange of CO$_2$ is peat loss due to the export of carbon in ground water as dissolved and particulate organic carbon. This is represented within $\epsilon$ in Eq.4 and is something that would be captured in subsidence studies. A recent paper (Cook et al., 2018) investigated fluvial carbon losses from study sites in the same plantation estates as our current study and reported losses of organic carbon in the range of 0.3 to 0.5 Mg ha$^{-1}$ yr$^{-1}$, this would add around 2.5% to our estimated peat organic carbon loss but would not raise it to the levels expected in Hooijer et al. (2012) or Page et al. (2011). Similarly, consideration needs to be given to potential emissions of soil carbon as methane (CH$_4$) even though, as reported in Couwenberg et al. (2009), CH$_4$ emissions from tropical peatlands are...
typically far lower than from boreal/temperate peatlands. Manning et al. (2019) monitored CH$_4$ emission from soils (and drainage channels) at the Sabaju and Sebungan plantations (though from different planting blocks to our OPnew and OPmature) and, as with their CO$_2$ results, found fluxes from Sabaju (0.03 Mg CH$_4$-C ha$^{-1}$ yr$^{-1}$) to be higher than from Sebungan (0.006 Mg CH$_4$-C ha$^{-1}$ yr$^{-1}$). While the GWP impact of methane is calculated at 34 times that of CO$_2$ (Myhre et al. 2013), in terms of soil carbon loss this level of carbon mobilisation would add only around 0.1% to our estimate of annual carbon loss over the entire chronosequence (increasing to around 0.3% if drainage water CH$_4$-C emissions were included).

Comparison to existing emission factors (as CO$_2$)

Our long term emission factor (calculated across all 151 months) for peat carbon loss (as CO$_2$) at 67 Mg CO$_2$ ha$^{-1}$ yr$^{-1}$ is closer to the IPCC emission factor for peatland conversion to acacia plantation, 73 Mg CO$_2$ ha$^{-1}$ yr$^{-1}$ than oil palm which is lower at 40 Mg CO$_2$ ha$^{-1}$ yr$^{-1}$ (Hiraishi et al., 2014). The effect of plantation species on peatland CO$_2$ emission was not found to be a significant factor in the study of Carlson et al. (2015), who discussed the likely importance of time since drainage, though their dataset was limited to a narrow age range. Miettinen et al. (2017) preferred to use the mean of the two IPCC factors, (55 Mg CO$_2$ ha$^{-1}$ yr$^{-1}$), in their calculation of carbon loss across the region due to peatland conversion which agrees well with the Cooper et al. (2020) mean figure of 53.1 Mg CO$_2$ ha$^{-1}$ yr$^{-1}$. All these estimates remain lower than the EPA accepted emission factor of 95 Mg CO$_2$ ha$^{-1}$ yr$^{-1}$ (EPA 2014). The IPCC explicitly exclude the first 6 years of conversion in their emission factor due to lack of data but acknowledge that this period would see much higher carbon losses. This is clearly demonstrated by our estimate of soil carbon emissions for years 1-6 at 91.6 Mg CO$_2$ ha$^{-1}$ yr$^{-1}$. The overall net emission of CO$_2$ to the atmosphere (NEE) for this period, accommodating forest biomass decomposition and photosynthetic uptake, was higher at 110.8 Mg CO$_2$ ha$^{-1}$ yr$^{-1}$.

This difference between direct soil carbon emission, ($\Delta$SOC, as CO$_2$), and the net ecosystem scale addition of CO$_2$ to the atmosphere (NEE), is an important distinction, particularly in the early years of conversion, and should be considered when assessing the impacts of land-use change. In Table 2, we present, based on our results, estimates of emission factors for both NEE and $\Delta$SOC at an annual timestep across a 12-year period. Taking the cumulative sum (for either
component) for any given period and dividing by that number of years will provide an estimate of mean annual CO₂ emission for that period.

Conclusions

Despite our results reporting lower peat carbon loss in the early years following conversion than subsidence studies might have suggested, there is no doubt that these emissions remain extremely significant and peat swamp forest conversion to agriculture results in very large net emissions of CO₂. We have shown that the impact of these fluxes on the atmospheric carbon pool can be larger than emission factors for soil carbon loss alone might suggest, and that is highly unlikely that ‘carbon debts’ incurred early in a plantation lifecycle could be recouped over the remaining years. Evidence has shown that moratoria on peatland conversion within protected areas can be effective (Chen et al., 2019) but huge challenges remain and newly identified areas of extensive peatland being reported from tropical zones across the globe (Lähteenoja and Page, 2011, Xu et al., 2018) reveal regions that may be particularly vulnerable to land-use change. Despite policy development in South East Asia, limitations in regulatory frameworks and enforcement capabilities combined with political and socio-economic factors still challenge peatland protection (Padfield et al., 2016, Wijedasa et al., 2018). Our results should make it clear that conservation of these globally important carbon stocks is vital to any efforts to minimise the impacts of future climate change and reduce the contribution of land-use change to it.

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**Competing interest declaration**

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**Figure Legends:**

**Figure 1:** Daily climate data from OPnew and OPmature. Plots (a) and (b) show rainfall (left Y axis) and water table depth (right Y axis) for each site with mean WTD indicated by a dashed red line. Plots (c) and (d) show air and soil temperature at each site, again with dashed red lines indicating the mean for each parameter over the study period.

**Figure 2:** Monthly total CO$_2$ flux for OPnew and OPmature partitioned into gross photosynthetic uptake (GPP) and ecosystem respiration ($R_{eco}$). X axis labels indicate sampling date, inset year labels indicate which growing year (post plantation establishment) each monitoring period overlaps.

**Figure 3:** Monthly chronosequence plot of net ecosystem exchange of carbon (NEE), combining OPnew and OPmature into a single timeseries. Blue solid line shows measured monthly sums of NEE from each site plotted against month since respective plantation establishment. Dotted grey line shows interpolated monthly values, green line shows NEE extrapolated for the first four months at OPnew, dashed red line shows cumulative NEE summed from the resulting complete timeseries. Positive values (i.e. $Y>0$) indicate a net emission of carbon from the ecosystem to the atmosphere, negative values a net uptake.

**Figure 4:** Recreation of Hooijer et al. (2010) linear fit of CO$_2$ emission to water table depth (closed symbols) with the inclusion of individual years from OPnew and OPmature (open symbols). Solid black line shows linear fit with intercept constrained to zero, dashed red line shows linear fit with intercept free. Note, the early conversion data (OPnew) are not included in this plot but their effect can be seen in supplementary materials, (Fig. S.1.5.) (units are presented as CO$_2$ as in the original Hooijer analysis, these may be converted to CO$_2$-C (carbon) through multiplying by a factor of 12/44)

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Figure 5: Relative change (%) in soil carbon emission following a change from an existing mean annual water table depth below the peat surface to a new target depth. Negative (green) values indicate a reduction, positive (orange) indicate an increase, ns (grey) values indicate predictions that are not statistically significant. For example, changing from 0.4 m WTD (horizontal row) to 0.3 m (vertical column) would result in a 10% reduction in soil carbon emission (intersect at -10).
Total flux (Mg CO$_2$ ha$^{-1}$ month$^{-1}$)
\[ y = 118x \quad R^2 = 0.85 \]
\[ y = 7.38 + 106x \quad R^2 = 0.34 \]
| Existing mean annual water table depth (m) | New mean annual water table depth (m) | \(0\) | \(0.1\) | \(0.2\) | \(0.3\) | \(0.4\) | \(0.5\) | \(0.6\) | \(0.7\) | \(0.8\) |
|-----------------------------------------|--------------------------------------|-------|-------|-------|-------|-------|-------|-------|-------|-------|
| \(0\)                                   | 0                                    | 0     | +38   | +71   | +99   | +120  | +137  | +148  | +153  | +154  |
| \(0.1\)                                 | -28                                  | 0     | +24   | +43   | +59   | +71   | +79   | +83   | +83   |        |
| \(0.2\)                                 | -42                                  | -19   | 0     | +16   | +29   | +38   | +45   | +48   | +48   |        |
| \(0.3\)                                 | -50                                  | -30   | -14   | 0     | +11   | +19   | +25   | +28   | +28   |        |
| \(0.4\)                                 | -55                                  | -37   | -22   | -10   | 0     | +7    | +12   | +15   | +15   |        |
| \(0.5\)                                 | -58                                  | -42   | -28   | -16   | -7    | 0     | +5    | +7    | +7    |        |
| \(0.6\)                                 | -60                                  | -44   | -31   | -20   | -11   | -4    | 0     | +2\textsuperscript{ns} | +2\textsuperscript{ns} |        |
| \(0.7\)                                 | -61                                  | -45   | -32   | -22   | -13   | -7    | -2\textsuperscript{ns} | 0     | 0\textsuperscript{ns} |        |
| \(0.8\)                                 | -61                                  | -45   | -33   | -22   | -13   | -7    | -2\textsuperscript{ns} | 0\textsuperscript{ns} | 0     |        |

*All crosswise comparisons are significantly different to each other at the 95% confidence interval except where stated as zero or not significant (ns)*