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Modeling relationships between water table depth and peat soil carbon loss in Southeast Asian plantations

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Abstract

Plantation-associated drainage of Southeast Asian peatlands has accelerated in recent years. Draining exposes the upper peat layer to oxygen, leading to elevated decomposition rates and net soil carbon losses. Empirical studies indicate positive relationships between long-term water table (WT) depth and soil carbon loss rate in peatlands. These correlations potentially enable using WT depth as a proxy for soil carbon losses from peatland plantations. Here, we compile data from published research assessing WT depth and carbon balance in tropical plantations on peat. We model net carbon loss from subsidence studies, as well as soil respiration (heterotrophic and total) from closed chamber studies, as a function of WT depth. WT depth across all 12 studies and 59 sites is 67 ± 20 cm (mean $\pm$ standard deviation). Mean WT depth is positively related to net carbon loss, as well as soil respiration rate. Our models explain 45% of net carbon loss variation and 45–63% of soil respiration variation. At a 70 cm WT depth, the subsidence model suggests net carbon loss of 20 tC ha^{-1} yr^{-1} (95% confidence interval (CI) 18–22 tC ha^{-1} yr^{-1}) for plantations drained for >2 yr. Closed chamber-measured total soil respiration at this depth is 20 tC-CO$_2$ ha^{-1} yr^{-1} (CI 17–24 tC-CO$_2$ ha^{-1} yr^{-1}) while heterotrophic respiration is 17 tC-CO$_2$ ha^{-1} yr^{-1} (CI 14–20 tC-CO$_2$ ha^{-1} yr^{-1}), ~82% of total respiration. While land use is not a significant predictor of soil respiration, WT depths are greater at acacia (75 ± 16 cm) than oil palm (59 ± 15 cm) sample sites. Improved spatio-temporal sampling of the full suite of peat soil carbon fluxes—including fluvial carbon export and organic fertilizer inputs—will clarify multiple mechanisms leading to carbon loss and gain, supporting refined assessments of the global warming potential of peatland drainage.

1. Introduction

High water tables (WTs) and low decomposition rates characterize peatlands. In Southeast Asia, these organic soils sequester ~68.5 Tg of carbon, representing 11–14% of the global peatland carbon pool [1]. Draining tropical peatlands leads to net soil carbon loss [2], and plantation agriculture is a leading cause of peatland drainage across Southeast Asia [3–5]. Industrial plantations including oil palm (*Elaeis guineensis*) and pulpwood (e.g., acacia (*Acacia crassicarpa*)), occupy ~20% of peatland area in Peninsular Malaysia, Sumatra, and Borneo [4]. The relative proportion of plantations on peatlands in Southeast Asia increased over the last 20 years, from 12% in 1990 to 18% in 2010 [6], generating substantial greenhouse gas (GHG) emissions from peat decomposition [3–5, 7–9].

WT depth—the distance between the ground surface and WT, with positive values indicating a WT below ground level—is an expression of the soil moisture profile. Soil moisture is recognized as a control on peat carbon loss because it mediates the volume of peat substrate exposed to oxygen, influencing microbial activity and decomposition [10–16]. Thus, soil moisture affects both methane (CH$_4$) and carbon

dioxide (CO_2) emissions from the land surface. Methane emissions are a function of anaerobic methanogenesis and CH_4 oxidation (aerobic and anaerobic), as well as transport mechanisms (e.g., diffusion, plant-mediated, ebullition) [17, 18]. In tropical systems, CH_4 emissions are related to WT depth [19, 20]. At depths >20 cm, studies frequently report near-zero CH_4 emissions or even net uptake from the atmosphere, although large net CH_4 emissions have also been recorded in drained systems [19–22]. At lower WT depths, net emissions are typical [19]; for example, the IPCC reports 0.041 tC-CH_4 ha^{-1} yr^{-1} in tropical forested land uses with <30 cm WT depth [2].

Empirical work in Southeast Asia suggests positive relationships between WT depth and soil CO_2 emissions from decomposition of organic matter. Couwenberg *et al* [19] compiled data from diverse tropical land uses and found that total soil respiration rates tend to be higher at greater WT depths. While Jauhiainen *et al* [24] found a positive linear correlation between mean WT depth and daytime heterotrophic ($r^2 = 0.47$) and total ($r^2 = 0.34$) soil respiration rates in a Sumatra pulpwood plantation, instantaneous WT depth was uncorrelated with CO_2 emissions [24]. This finding suggests that relationships between soil respiration and WT depth may only be detectable when measured over long time frames. Moreover, this relationship is likely to be nonlinear, especially at high or low WT depths [11, 19, 23–25]. For example, Hirano *et al* [23] used chamber gas flux measurements collected at forest and agricultural sites in Kalimantan to develop nonlinear models predicting ecosystem respiration from WT depth and maximum CO_2 flux ($r^2 = 0.28$–0.83).

WTs also influence other pathways of peat carbon loss. Drainage affects export of fluvial dissolved and particulate carbon, which are produced in soil and then transported to the drainage network, where they may be emitted to the atmosphere as CH_4 or CO_2, or deposited in wetlands, estuaries, or oceans [26–29]. Net carbon losses from peat decomposition may also be correlated with WT depth. Hooijer *et al* [30] found that total net peat carbon loss is positively related to long term mean WT depth under acacia plantations, while suggesting that other factors (e.g., fertilizer inputs) may sometimes override the effects of WT depth on carbon losses. Fires are rare in inundated peat swamps but prevalent in drained peat, and lead to substantial carbon emissions [2, 31–36]. Peatlands adjacent to drained areas may also experience reduced WTs and associated carbon loss [5, 30]. In sum, tropical peatland drainage leads to substantial on- and off-site carbon losses via increased decomposition rates, changes in fluvial transport, and higher fire risk.

Acknowledging the substantial carbon losses associated with peatland drainage, commodity companies are attempting to reduce peat-related carbon emissions in their supply chains [e.g., 37], jurisdictions are quantifying peat carbon fluxes for REDD + initiatives

[38], and certification organizations are requiring peat GHG accounting [39]. Emissions models that incorporate management factors—including WT depth—potentially support such site-specific assessments of carbon loss from plantations on peatlands. Yet in industrial plantations, relationships between WT depth and carbon loss have been unclear, and the use of WT depth as a proxy for carbon loss remains contentious [40–43].

Recognizing the need for inter-site assessments of carbon loss-WT depth relationships, here we compile and analyze data from published research reporting WT depth and total carbon loss or soil respiration in tropical peatlands drained for plantation land use. We aim to answer the following questions: (1) what methods are commonly used to quantify carbon loss from tropical peatlands and what are the benefits and drawbacks of these methods? (2) Is carbon loss and/or soil respiration related to WT depth across diverse sites? (3) If so, what is the functional form and strength of these relationships? (4) How do managerial and environmental factors (e.g., vegetation age) affect carbon loss rates?

2. Quantifying net soil carbon loss from peat drainage

Estimating soil carbon loss from plantations on peatlands requires calculating the difference in carbon balance between non-plantation and plantation ecosystems:

$$f_{net} = f_{plantation} - f_{initial} \qquad (1)$$

where $f_{plantation}$ is net peat soil carbon flux in a plantation (C ha^{-1} yr^{-1}), $f_{initial}$ is net peat soil carbon flux from the pre-plantation land use, and f_{net} is net peat soil carbon loss from plantation establishment.

Here, we briefly describe two techniques—soil subsidence and mass balance—that have been used to estimate net soil carbon loss from tropical peatlands cleared for plantations. Details are provided in the supplementary text. Our analysis excludes carbon fluxes from live biomass change, peat burning, and plantation activities such as palm oil wastewater effluent management.

2.1. Subsidence

Subsidence measurements, and information about the WT and peat itself, permit net carbon loss assessments [30, 43–45]. Models assume that subsidence results from three processes: (1) decomposition, carbon loss from peat soil organic matter (SOM); (2) compaction and shrinkage, peat volume reduction above the WT; and (3) consolidation, compression of saturated peat below the WT.

Subsidence models integrate all forms of SOM carbon loss and gain, and do not require expensive equipment. Yet, subsidence models cannot allocate carbon loss among different pathways (e.g., fluvial, soil

surface) and species (e.g., CO_2, CH_4). Methane has a global warming potential 34 times greater than CO_2 over a 100 yr time frame [46]. If much carbon is emitted as CH_4, the contribution of peatland drainage to radiative forcing will be greater; if much carbon is stored in wetlands, estuaries, and oceans, the contribution will be less. Thus, subsidence models alone cannot inform the global warming potential of peatland drainage.

2.2. Mass balance

Mass balance methods estimate f by accounting for soil carbon losses and gains. Models must consider many pathways and species of carbon [47]. With this in mind, we present a model estimating $f_{plantation}$ in terms of carbon pathways:

$$f_{plantation} = \left(C_H + C_{CH_4} + C_{H_2O} \right) \\ - \left(C_{GPP} + C_F + C_D + C_W \right). \quad (2)$$

In this model, C_H is soil CO_2 emissions from heterotrophic respiration. Total soil respiration (C_S) measured with closed chambers consists of carbon loss from microbial decomposition of SOM and dead plant remains (C_H), plus SOM-derived emissions due to the rhizosphere priming effect, rhizomicrobial respiration, and root respiration [48]. C_{CH_4} is soil CH_4 emissions; C_{H_2O} is fluvial export of carbon derived from SOM, including dissolved organic carbon (DOC), particulate organic carbon, and dissolved inorganic carbon; C_{GPP} is carbon input from gross primary production, including leaf litter, root exudates, and root mortality; C_F is carbon input from organic fertilizer; C_D is carbon input from atmospheric deposition; and C_W is carbon input from weathering of underlying strata.

The mass balance approach elucidates mechanisms contributing to carbon loss and gain, improving scientific understanding of the processes driving carbon flux. By distinguishing between GHG species, it permits accurate global warming potential assessments. In practice, however, measuring these quantities can be experimentally challenging [48, 49]. No tropical plantation peat study has measured all fluxes in a single location, and data are sparse except for C_H and C_{CH_4}. Only a few studies [27, 28, 50, 51] report C_{H_2O} and C_{GPP} in plantations on tropical peatlands (supplementary text). Comprehensive estimates of C_F are not available, and fertilizer inputs likely vary widely among sites [52]. Although C_D could be important in Southeast Asia due to fire-associated organic and black carbon deposition [53, 54], no studies estimate carbon atmospheric deposition rates in tropical peatlands. Mass balance components also vary greatly in magnitude. In drained tropical peatlands, research suggests that C_{CH_4} is small (<1% in carbon equivalence) compared to C_H [19, 20, 49]. Research in Great Britain suggests that C_W is <4% of C_S [55]. Considering data availability and the relative magnitude of each mass

balance component, the following analysis focuses on C_H.

3. Data collection

We conducted a comprehensive literature search to generate papers reporting WT depth and carbon loss from tropical peatlands, and then applied a set of criteria to select studies for analysis (database 1, supplementary text). Twelve studies—four subsidence, eight soil respiration from closed chambers—are included in our analysis (database 2). From these studies, a total of 59 sites were selected. We defined a 'site' as a unique management regime, vegetation age, or plant species, or a data point representing >2 spatial replicates and reported separately in original manuscript. For each site, we derived the mean and standard deviation of WT depth and soil CO_2 emissions and/or carbon loss from subsidence (supplementary text). All studies measured WT depth as the distance between soil surface and groundwater level (see database 2 for measurement methods). Since groundwater depths are less than WT depths at field collection drains [44, 56, 57], practitioners should apply our models only to groundwater measurements.

Only two subsidence studies [30, 58] met all criteria. To generate a larger sample size, we applied our own calculations to generate carbon loss from Othman *et al* [59] and DID and LAWOO [60]. Othman *et al* [59] do not clearly describe sampling methods. Moreover, dry bulk density (DBD, $g\,cm^{-3}$) appears to be based on a conceptual model instead of field measurements. To generate data from this study, we excluded measurements collected 0–5 yr after plantation establishment when carbon loss and subsidence rates may be elevated [30]. We grouped subsidence measurements into shallow, medium, and deep peat (table 1), and calculated weighted average subsidence and WT depth in these groups. DID and LAWOO [60] did not measure DBD. We applied the DBD values suggested by Couwenberg and Hooijer [58] to Othman *et al* [59] and DID and LAWOO [60] (0.08 $g\,cm^{-3}$, and 0.07 $g\,cm^{-3}$, respectively), and assumed a DBD standard deviation of 50%. Our need to apply such assumptions highlights the lack of subsidence-based carbon loss studies in plantations.

4. Statistical models

We evaluated linear (first order polynomial) and nonlinear (second order polynomial) models relating carbon loss ($tC\,ha^{-1}\,yr^{-1}$) to WT depth (cm). We also considered land use, mean vegetation age, mean peat depth, and mean time since draining as additional independent variables. Using the Akaike information criterion corrected for small sample sizes (AICc) to gauge relative model quality [61], we found that linear models outperformed nonlinear options, likely due to

Table 1. Site data derived in subsidence studies in Southeast Asian plantations on drained peatlands. The number of samples represents the number of subsidence poles at a site. We applied dry bulk density values suggested by Couwenberg and Hooijer [58] to DID and LAWOO [60] and Othman *et al* [59] studies, and assumed 50% standard deviation. Jambi and Riau are located in Sumatra, Indonesia; Sarawak is in Malaysian Borneo; and Johor is in Peninsular Malaysia. Values are reported as mean ± standard deviation.

Study	Location	Veg.	Sample Site	Time Since Draining (yrs)	Mean Peat Depth (m)	Veg. Age (yrs)	Sample Period (yrs)	Samples (#)	Water table (cm)	Subsidence (cm yr^{-1})	Dry Bulk Density (g cm^{-3})	Carbon Density (gC cm^{-3})	Carbon Loss (tC ha^{-1} yr^{-1})
Couwenberg and Hooijer 2013	Jambi	Oil palm	19OP	19 (15–20)	7.7	16–19	3	34	65 ± 25	3.7 ± 0.50	0.078 ± 0.010	0.043 ± 0.0082	16 ± 3.7
	Jambi	Oil palm	5OP	5 (4–7)	6.3	4–7	3	17	56 ± 6.0	3.9 ± 0.50	0.082 ± 0.010	0.045 ± 0.0082	18 ± 3.9
DID and LAWOO 1996	Johor	Oil palm	1, 7, 11, 20, 21, 30–33	>28	8.5	n/a	4	9	53 ± 16	3.8 ± 0.86	0.070 ± 0.035	0.039 ± 0.021	15 ± 8.6
Hooijer *et al* 2012	Riau	Acacia	A	6 (3–8)	9.0	0–5	2	6	56 ± 11	5.9 ± 1.9	0.074 ± 0.020	0.041 ± 0.013	24 ± 11
	Riau	Acacia	B	6 (3–8)	9.0	0–5	2	6	63 ± 10	5.2 ± 2.1	0.074 ± 0.020	0.041 ± 0.013	21 ± 11
	Riau	Acacia	C	6 (3–8)	9.0	0–5	2	6	54 ± 11	4.5 ± 2.1	0.074 ± 0.020	0.041 ± 0.013	19 ± 11
	Riau	Acacia	D	6 (3–8)	9.0	0–5	2	5	72 ± 10	5.7 ± 2.4	0.074 ± 0.020	0.041 ± 0.013	23 ± 12
	Riau	Acacia	E	6 (3–8)	9.0	0–5	2	6	84 ± 10	5.6 ± 2.1	0.074 ± 0.020	0.041 ± 0.013	23 ± 11
	Riau	Acacia	F	6 (3–8)	9.0	0–5	2	6	56 ± 11	4.0 ± 1.9	0.074 ± 0.020	0.041 ± 0.013	16 ± 9.5
	Riau	Acacia	G	6 (3–8)	9.0	0–5	2	5	43 ± 11	3.4 ± 2.0	0.074 ± 0.020	0.041 ± 0.013	14 ± 9.4
	Riau	Acacia	H	6 (3–8)	9.0	0–5	2	5	42 ± 11	2.9 ± 1.5	0.074 ± 0.020	0.041 ± 0.013	12 ± 7.3
	Riau	Acacia	I	6 (3–8)	9.0	0–5	2	5	67 ± 11	3.8 ± 1.4	0.074 ± 0.020	0.041 ± 0.013	16 ± 7.7
	Riau	Acacia	K	6 (3–8)	9.0	0–5	2	5	74 ± 11	4.7 ± 1.7	0.074 ± 0.020	0.041 ± 0.013	19 ± 9.4
	Riau	Acacia	L	6 (3–8)	9.0	0–5	2	5	61 ± 12	3.1 ± 1.7	0.074 ± 0.020	0.041 ± 0.013	13 ± 8.1
	Riau	Acacia	M	6 (3–8)	9.0	0–5	2	5	52 ± 13	3.5 ± 1.9	0.074 ± 0.020	0.041 ± 0.013	14 ± 9.1
	Riau	Acacia	N	6 (3–8)	9.0	0–5	2	5	74 ± 10	5.4 ± 1.8	0.074 ± 0.020	0.041 ± 0.013	22 ± 10
	Riau	Acacia	O	6 (3–8)	9.0	0–5	2	9	71 ± 12	5.8 ± 1.9	0.074 ± 0.020	0.041 ± 0.013	24 ± 11
	Riau	Acacia	P	6 (3–8)	9.0	0–5	2	5	88 ± 13	5.3 ± 2.0	0.074 ± 0.020	0.041 ± 0.013	22 ± 11
	Riau	Acacia	Q	6 (3–8)	9.0	0–5	2	5	73 ± 13	3.3 ± 2.3	0.074 ± 0.020	0.041 ± 0.013	14 ± 10
	Riau	Acacia	R	6 (3–8)	9.0	0–5	2	6	69 ± 21	4.0 ± 2.4	0.074 ± 0.020	0.041 ± 0.013	16 ± 11
	Riau	Acacia	S	6 (3–8)	9.0	0–5	2	5	73 ± 26	7.4 ± 2.4	0.074 ± 0.020	0.041 ± 0.013	30 ± 14
	Riau	Acacia	T	6 (3–8)	9.0	0–5	2	7	75 ± 26	7.3 ± 2.4	0.074 ± 0.020	0.041 ± 0.013	30 ± 14
	Riau	Acacia	U	6 (3–8)	9.0	0–5	2	7	93 ± 21	6.4 ± 2.2	0.074 ± 0.020	0.041 ± 0.013	26 ± 12
	Riau	Acacia	V	6 (3–8)	9.0	0–5	2	6	108 ± 22	5.9 ± 2.2	0.074 ± 0.020	0.041 ± 0.013	24 ± 12
Othman *et al* 2011	Sarawak	Oil palm	deep	>5	2.5	5–9	7	18	45 ± 4.2	4.1 ± 1.5	0.080 ± 0.040	0.044 ± 0.024	18 ± 12
	Sarawak	Oil palm	moderate	>5	2.5	6–13	7	2	33 ± 1.9	2.8 ± 1.7	0.080 ± 0.040	0.044 ± 0.024	12 ± 10
	Sarawak	Oil palm	shallow	>5	2.5	6–13	7	5	37 ± 1.6	2.4 ± 1.2	0.080 ± 0.040	0.044 ± 0.024	11 ± 7.7

no observations at low (<20 cm) and high (>110 cm) WT depths. Thus, we present only linear models here.

Subsidence models assess net carbon loss (f), which is distinct from heterotrophic respiration (C_H) and total soil respiration (C_S). The number of samples per site varied greatly, and large sample sizes exerted overbearing influence on regressions. We present non-weighted models in the main text, and report weighted models in table S1 and figures S1–S2. Analysis was performed in R [62]. We report the standard deviation of the sample mean.

5. Carbon loss from subsidence studies

Four subsidence studies delivered 27 sites in oil palm and acacia plantations in Indonesia and Malaysia (table 1, figure 1). The Riau acacia and Jambi oil palm sites include subsidence measurements 3–4 yr post-draining, when decomposition and subsidence rates may be elevated [16, 30]. Yet, subsidence rates at these sites were similar to those measured in sites >5 yr post-draining (table 1). Mean WT depth across sites was 64 ± 17 cm. The three Sarawak oil palm sites were part of a drainage experiment that produced unusually low WT depths (mean 39 cm) [59]. While such depths should not be considered representative of typical oil palm plantation drainage practices, these samples clarify the relationship between low WT depth and carbon loss.

Subsidence-measured net carbon loss is positively related to WT depth (table 3), and WT depth is a significant predictor of carbon loss ($p < 0.01$, $r^2 = 0.45$). No additional predictor variables (mean vegetation age, land use, mean peat depth, and mean time since draining) were significant when included along with WT depth ($p > 0.05$). At 70 cm WT depth—chosen for comparative purposes with other studies that also report emissions at this depth—we estimate net carbon loss of 20 tC ha^{-1} yr^{-1} (confidence interval (CI) 18–22 tC ha^{-1} yr^{-1}) for areas drained for >2 yr. Excluding the problematic Othman *et al* [59] study from regressions changed carbon loss predictions by <1% at 70 cm WT depth.

6. Soil respiration from closed chamber studies

Eight chamber-based studies meet quality benchmarks, and provide 32 sites, including 16 total and 16 heterotrophic respiration sites (table 2, figure 2). Studies were conducted in oil palm, acacia, melaleuca (*Melaleuca sp.*), rubber (*Hevea brasiliensis*), and sago (*Metroxylon sagu*) plantations in Indonesia and Malaysia. Sites were drained >5–16 yr prior to measurements and vegetation was <1–14 yr old. WT depth was 66 ± 23 cm at total respiration sites, and 72 ± 20 cm at heterotrophic respiration sites. Most studies measured soil CO_2 flux during daytime hours, when soil temperature tends to be higher than at night, leading to inflated emission estimates [23, 24, 63]. An exception, Marwanto and Agus [64] assessed emissions throughout the day and night. To correct for elevated daytime soil respiration rates, we applied the 14.5% reduction suggested by Jauhiainen *et al* [24] to all sites except Marwanto and Agus [64]. This correction is based on a single study and is therefore uncertain. We present uncorrected respiration rates in table 2.

Positive relationships between WT depth and soil respiration rates were detected across all land uses ($p < 0.01$, $r^2 = 0.45$ (C_H) and 0.56 (C_S), table 3). At 70 cm WT depth, heterotrophic emissions are estimated at 17 tC-CO_2 ha^{-1} yr^{-1} (CI 14–20 tC-CO_2 ha^{-1} yr^{-1}), ~82% of total respiration emissions (20 tC-CO_2 ha^{-1} yr^{-1}, CI 17–24 tC-CO_2 ha^{-1} yr^{-1}). Mean vegetation age is a significant predictor of heterotrophic respiration but not total respiration in a model including WT depth (table 3). Given the limited range of vegetation ages across sites (figure 3, table 2, figure S3), the model is applicable only to vegetation ages <6 yr old. Land use, mean peat depth, and mean time since draining were not significant predictors of respiration rates when included in models with WT depth ($p > 0.05$, figure S3).

7. Comparison with other estimates

Our subsidence model estimate of 20 tC ha^{-1} yr^{-1} at 70 cm WT depth is similar to total carbon loss presented by other authors [58, 65] (table 4). Our model has slope $\beta = 0.21$, comparable to that of Hooijer *et al* [30] ($\beta = 0.19$), although intercept terms differ ($\alpha = 5.4$ in our model versus 0.057 for [30]). Published acacia emissions factors are also similar to our model estimates (table 4). Yet, our estimate differs from oil palm emissions factors presented by Murdiyarso *et al* [9], Hergoualc'h and Verchot [49], and the IPCC [2], who report losses of 5.2 ± 1.1, 8.2 ± 2.9, and 12 tC ha^{-1} yr^{-1} (after addition of a DOC emission factor), respectively.

Hergoualc'h and Verchot [49] update and refine the work of Murdiyarso *et al* [9], and use similar emission factors to the IPCC approach. Thus, we investigated the model presented by Hergoualc'h and Verchot [49], who specify a WT depth of 66 ± 13 cm. Their mass balance model includes: (1) C_{GPP} comprising litterfall (1.5 ± 0.1 tC ha^{-1} yr^{-1}) and root mortality (3.6 ± 1.1 tC ha^{-1} yr^{-1}); (2) C_{H_2O} in the form of fluvial DOC export (0.9 ± 0.1 tC ha^{-1} yr^{-1}); and (3) C_H of 12 ± 2.7 tC-CO_2 ha^{-1} yr^{-1}, derived by applying a 73 ± 13% correction factor to total soil respiration (C_S). Plugging our predicted heterotrophic respiration rate of 16 tC-CO_2 ha^{-1} yr^{-1} at 66 cm WT depth into this mass balance model, we estimate 12 tC ha^{-1} yr^{-1} (CI 8–15 tC ha^{-1} yr^{-1}) of net carbon loss. This is 47–70% of the 19 tC ha^{-1} yr^{-1} (CI 18–21 tC ha^{-1} yr^{-1}) carbon loss rate estimated by our subsidence model at

Table 2. Studies and sites used to build models of total (T) and heterotrophic (H) soil respiration in Southeast Asian plantations on drained peatlands. Respiration rates are not temperature corrected. The number of samples represents the replicates in space multiplied by the samples in time. Jambi and Riau are located in Sumatra, Indonesia, and Sarawak is in Malaysian Borneo. Values are reported as mean ± standard deviation.

Study	Type	Location	Veg.	Sample Site	Sample Location/Root Exclusion	Time Since Draining (yrs)	Mean Peat Depth (m)	Veg. Age (yrs)	Sample Period (yrs)	Samples (#)	Water table (cm)	Respiration (tC-CO_2 ha^{-1} yr^{-1})
Comeau et al 2013	H	Jambi	Oil palm	Long-term	between trees	7	6.0	5.0	0.75	146	75 ± 11	31 ± 18
	T	Jambi	Oil palm	Long-term	1 m	7	6.0	5.0	0.75	146	81 ± 12	36 ± 8.8
Dariah et al 2013	H	Jambi	Oil palm	Arang-Arang	>2.5 m	7	2.8	6.0	0.92	120	52 ± 21	10 ± 2.6
	T	Jambi	Oil palm	Arang-Arang	⩽2.5 m	7	2.8	6.0	0.92	160	56 ± 25	12 ± 3.1
Husnain et al 2014	H	Riau	Acacia	R-Ac-3	between trees (⩽1 m)	6	5.5	3.0	1.0	50	81 ± 24	16 ± 5.2
	H	Riau	Oil palm	R-OP-4	>3 m	6	5.5	4.0	1.0	50	72 ± 37	18 ± 6.8
	H	Riau	Rubber	R-Rb-6	between trees	6	5.5	6.0	1.0	50	67 ± 25	14 ± 4.6
Jauhiainen et al 2012	H	Riau	Acacia	A	>1 m	7	4.4	2.6	2.0	158	92 ± 27	27 ± 7.0
	H	Riau	Acacia	B	>1 m/trenched	7	8.4	2.7	2.0	192	78 ± 15	25 ± 8.7
	H	Riau	Acacia	C	>1 m	7	4.9	3.9	2.0	140	103 ± 32	28 ± 12
	H	Riau	Acacia	D	>1 m/trenched	7	4.9	0.11	2.0	350	69 ± 25	22 ± 6.0
	H	Riau	Acacia	E	>1 m	7	5.3	0.50	2.0	34	75 ± 23	19 ± 9.4
	H	Riau	Acacia	F	>1 m	7	8.7	0.59	2.0	74	85 ± 14	26 ± 10
	H	Riau	Acacia	H	>1 m/trenched	7	5.6	1.0	2.0	127	93 ± 16	38 ± 12
	H	Riau	Melaleuca	G	>1 m/trenched	7	5.6	1.0	2.0	143	45 ± 12	20 ± 6.8
	T	Riau	Acacia	A	<1 m	7	4.4	2.6	2.0	284	94 ± 28	42 ± 16
	T	Riau	Acacia	B	<1 m	7	8.4	2.7	2.0	270	73 ± 17	30 ± 10
	T	Riau	Acacia	C	<1 m	7	4.9	3.9	2.0	222	108 ± 33	31 ± 14
	T	Riau	Acacia	D	<1 m	7	4.9	0.11	2.0	71	78 ± 24	17 ± 7.6
	T	Riau	Acacia	E	<1 m	7	5.3	0.50	2.0	44	70 ± 22	18 ± 11
	T	Riau	Acacia	F	<1 m	7	8.7	0.59	2.0	154	84 ± 14	33 ± 11
	T	Riau	Acacia	H	<1 m	7	5.6	1.0	2.0	29	86 ± 12	32 ± 10
	T	Riau	Melaleuca	G	<1 m	7	5.6	1.0	2.0	30	36 ± 9.0	21 ± 6.5
Marwanto and Agus 2013	H	Jambi	Oil Palm	Sumber Agung	between trees	16	5.5	14	1.0	480	91 ± 14	13 ± 8.2
Melling et al 2013a	H	Sarawak	Oil palm	oil palm	canopy edge/excised	5	5.6	4.0	1.0	36	58 ± 9.0	6.9 ± 3.3
	H	Sarawak	Sago palm	sago palm	canopy edge/excised	5	6.5	4.0	1.0	36	24 ± 11	7.6 ± 4.0
	T	Sarawak	Oil palm	oil palm	canopy edge	5	5.6	4.0	1.0	36	58 ± 9.0	18 ± 6.9
	T	Sarawak	Sago palm	sago palm	canopy edge	5	6.5	4.0	1.0	36	24 ± 11	16 ± 7.5
Melling et al 2013b	T	Sarawak	Oil palm	S1	n/a	9–11	5.2	1.0	2.0	24	56 ± 8.7	16 ± 2.5
	T	Sarawak	Oil palm	S2	n/a	9–11	4.8	5.0	2.0	24	67 ± 13	17 ± 2.7
	T	Sarawak	Oil palm	S3	n/a	9–11	3.8	7.0	2.0	24	56 ± 12	19 ± 2.4
Wantanabe et al 2009	T	Riau	Sago palm	P3	1.5 m	>5	n/a	6.0	2.5	32	34 ± 21	6.0 ± 2.4

Table 3. Linear models relating total peat soil carbon loss (f, tC ha^{-1} yr^{-1}), heterotrophic soil respiration (C_H, tC-CO$_2$ ha^{-1} yr^{-1}), and total soil respiration (C_S, tC-CO$_2$ ha^{-1} yr^{-1}) to water table depth (WT, cm) and vegetation age (age, years) for plantations on drained peatlands in Southeast Asia. We report the measurement type, number of site data points (n), model specification, coefficient of determination (r^2), and AICc, as well as coefficient values, standard errors (SE), and p-values (p). We do not present subsidence and total respiration models with vegetation age because age is not a significant parameter ($p > 0.40$). All models are significant ($p < 0.01$).

Type	n	Model	r^2	AICc	α	SE	p	β	SE	p	γ	SE	p
Subsidence	27	$f = \alpha + \beta$WT	0.45	158	5.4	3.1	0.096	0.21	0.047	<0.01			
Heterotrophic	16	$C_H = \alpha + \beta$WT	0.45	107	−0.47	5.4	0.93	0.25	0.072	<0.01			
Respiration		$C_H = \alpha + \beta$WT $+ \gamma$Age	0.63	104	2.2	4.7	0.65	0.26	0.061	<0.01	−0.94	0.37	0.026
Total Respiration	16	$C_S = \alpha + \beta$WT	0.56	107	1.1	4.6	0.82	0.28	0.065	<0.01			

the same depth. Carbon input from root mortality is an extremely uncertain component of C_{GPP} [40]. By excluding root mortality, we estimate net carbon loss of 15 tC ha^{-1} yr^{-1} (CI 12–18 tC ha^{-1} yr^{-1}), 67–87% of our subsidence estimate. Considering that these carbon loss estimates were derived from completely independent methods and datasets, such convergence is striking. In the following section, we explore the factors—beyond the documented difference in heterotrophic respiration rates between models—that may account for remaining discrepancies between subsidence and mass balance models.

7.1. Subsidence model sources of uncertainty

Subsidence varies substantially across the land surface, and sufficient spatial sampling is required to reduce these uncertainties. For example, in Peninsular Malaysia, Wösten *et al* [44] report accumulated subsidence ranging from ∼20 to 140 cm across 17 locations. Peat substrate swells and shrinks depending on water inputs [66] and subsidence measurements must therefore account for temporal variation in precipitation. Additionally, most subsidence models must correctly partition among decomposition, compaction, and consolidation, while carbon content and DBD vary along the peat profile and also in space, and are not simple to measure [58, 67]. Page *et al* [65] emphasize the critical nature of these variables by re-parameterizing Couwenberg *et al*'s [19] subsidence-based model with alternate values for carbon density (gC cm^{-3}) and percent decomposition, generating net carbon losses ranging from 12 to 36 tC ha^{-1} yr^{-1}. While the model we used to assess subsidence carbon loss does not require partitioning among processes governing subsidence, it assumes dynamic equilibrium of the upper peat layer [58, 68]. If this assumption is not met, applying this model will misestimate carbon losses. Finally, in our analysis, the low number of available subsidence studies, and assumptions about DBD and carbon content values, generated predictions that were quite uncertain. Nevertheless, uncertainties surrounding carbon loss derived from subsidence measures are quantifiable because model parameters have well defined ranges, providing bounds on net carbon loss.
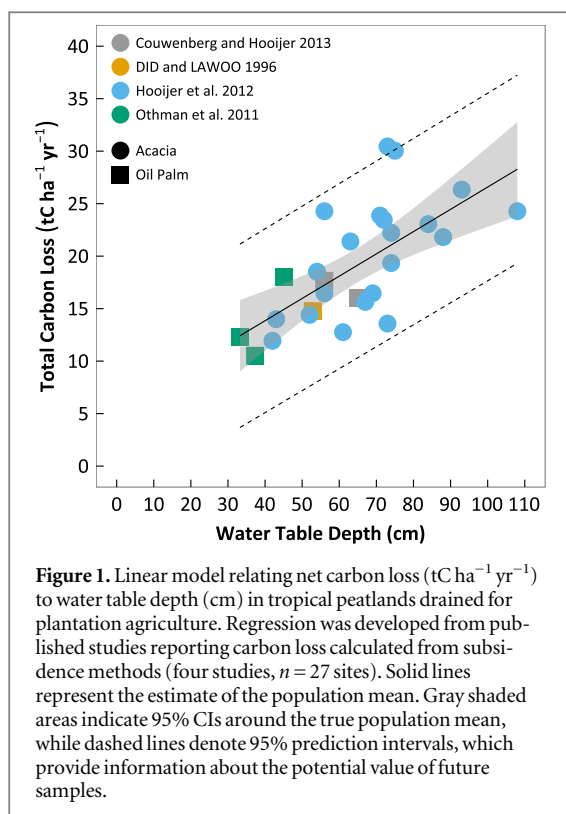
7.2. Mass balance model sources of uncertainty

Any mass balance formulation should include all forms of fluvial carbon loss, as well as carbon inputs via soil amendments, because these fluxes may be substantial. For example, in canals draining a Sumatra pulp plantation, water surface emissions were ∼15 tC-CO$_2$ and 2.4 tC-CH$_4$ ha^{-1} yr^{-1} [28]. Moreover, non-respiration carbon fluxes are likely to vary with WT depth. For instance, fluvial carbon export rates are a function not only of decomposition rates, but also fluvial transport, which is controlled by peat hydrology [69]. At present, the lack of published data describing C_{H_2O}, C_{GPP}, and C_F severely limits mass balance modeling efforts.

Accurately accounting for spatial and temporal heterogeneity requires detailed mass balance modeling. If soil carbon inputs are not assessed at the same site as losses, mass balance models may produce large errors. For instance, we question whether accounting for root carbon inputs is appropriate when using heterotrophic soil respiration measurements that exclude roots from sampling sites. Since soil carbon flux measured with closed chamber methods may have decomposed at another location [2, 17], lateral transport of carbon within the peat substrate is also important. Plantations include surfaces such as roads and drainage canals which remain under-sampled [70], and scaling point measurements up to plantations requires weighting fluxes by the area under each condition [2, 71]. Vegetation density varies between sites (e.g., oil palm densities in the present study range from 120 to 160 trunks ha^{-1}), influencing the quantity and distribution of litter and root inputs. Currently available data also limit mass balance model capacity to integrate temporal variation (e.g., leaf litter input changes with plantation maturation). Empirical research is needed to resolve spatio-temporal sampling limitations and to provide more data on fluxes beside soil respiration. Pending such research, the error associated with mass balance estimates is unknown.
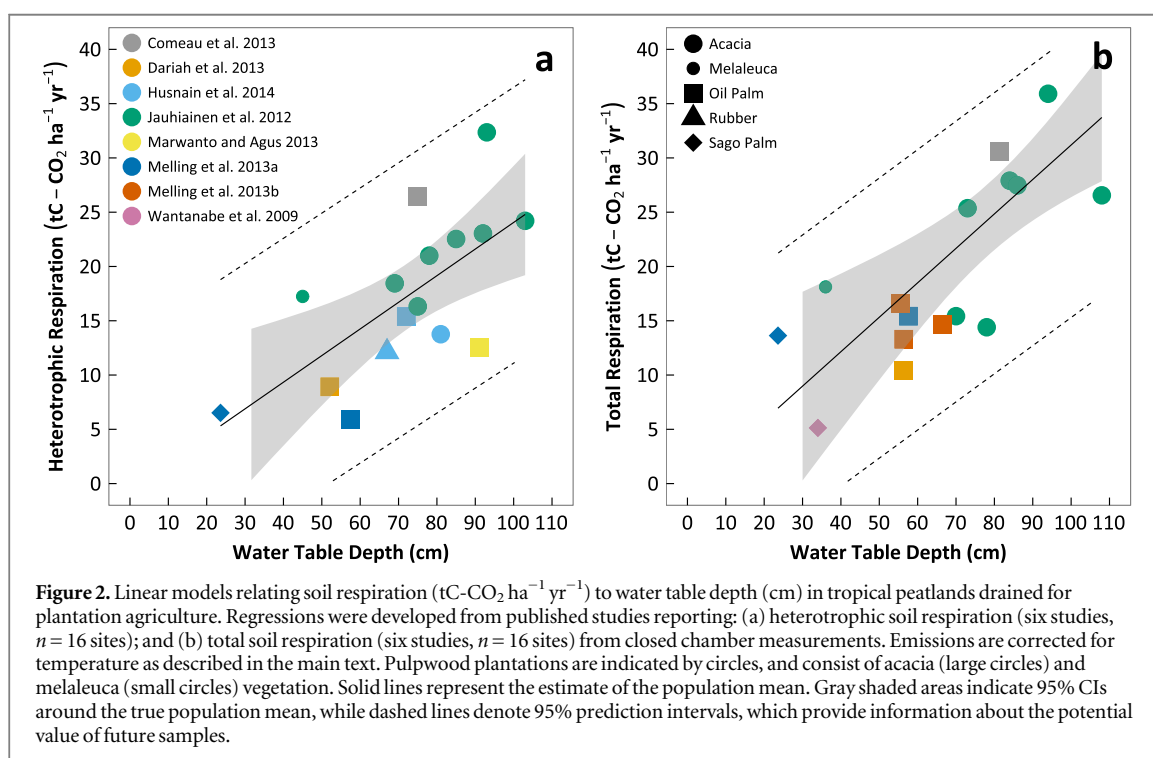
8. Confounding factors

Our simple WT models explain 45% of net carbon loss variation and 45–56% of soil respiration variation.

Figure 1. Linear model relating net carbon loss (tC ha^{-1} yr^{-1}) to water table depth (cm) in tropical peatlands drained for plantation agriculture. Regression was developed from published studies reporting carbon loss calculated from subsidence methods (four studies, $n = 27$ sites). Solid lines represent the estimate of the population mean. Gray shaded areas indicate 95% CIs around the true population mean, while dashed lines denote 95% prediction intervals, which provide information about the potential value of future samples.

may influence decomposition rates [23, 63, 72]. Yet the samples used to build our respiration models are skewed toward young vegetation ages (mean 3.4 yr, figure 3, figure S3), and we are uncertain if this relationship would hold over a more representative age sample. In contrast, land use is not a significant predictor of soil respiration when included with WT depth ($p = 0.52$ and 0.53 for C_S and C_H, respectively). Similarly, Page *et al* [65] propose that peat decomposition processes are similar in oil palm and acacia plantations. Nevertheless, factors such as litter quality and canopy cover, which alter decomposition dynamics [23, 63, 72], are likely to vary with vegetation type. While mean time since draining was also not a significant predictor of carbon loss when included in models along with WT depth ($p = 0.32$ and 0.25 for C_S and C_H, respectively, figure S3), most sites were just ~5–7 yr post-draining. In reality, peat soil carbon is lost rapidly in the first years after draining [15, 30, 44]. For example, Hooijer *et al* [30] report that carbon loss from an acacia plantation in the first 1–4 yr post-draining was 262% greater than 5–8 yr post-draining.

Fertilizer and soil disturbance are important sources of variation not included in our models due to

Figure 2. Linear models relating soil respiration (tC-CO$_2$ ha^{-1} yr^{-1}) to water table depth (cm) in tropical peatlands drained for plantation agriculture. Regressions were developed from published studies reporting: (a) heterotrophic soil respiration (six studies, $n = 16$ sites); and (b) total soil respiration (six studies, $n = 16$ sites) from closed chamber measurements. Emissions are corrected for temperature as described in the main text. Pulpwood plantations are indicated by circles, and consist of acacia (large circles) and melaleuca (small circles) vegetation. Solid lines represent the estimate of the population mean. Gray shaded areas indicate 95% CIs around the true population mean, while dashed lines denote 95% prediction intervals, which provide information about the potential value of future samples.

Factors beyond mean WT depth substantially influence soil carbon losses. Mean vegetation age is a significant predictor of heterotopic soil respiration (table 3). Respiration rates decline with increasing vegetation age, which is correlated with canopy cover and vegetative ground cover quality (figure S3). These factors affect soil temperature and litter inputs, which

inconsistent reporting across studies. Across our sites, nitrogen was applied 1–2 times yr^{-1} to oil palm, acacia was fertilized only during planting, and sago and rubber received no fertilizer. Nitrogen plays a regulatory role in CH$_4$ consumption [73], and has been linked to elevated soil CO$_2$ emissions in tropical drained peatlands [63]. Soil disturbance and compaction affect the

Table 4. Comparison of net carbon loss estimates from drained peatlands in Southeast Asia computed by mass balance (MB) and subsidence (S) models. For the IPCC estimates, we added the Tier I DOC loss (0.82 tC ha^{-1} yr^{-1}, 95% CI 0.56–1.14 tC ha^{-1} yr^{-1}) to Tier I CO$_2$ emission factors for oil palm and short-rotation plantations to derive total carbon loss. The Page *et al* (2011) rate was annualized over 50 years. Where available, standard deviation ($\pm$) or range (*x*–*x*) is reported.

| Study	Land use	Model type	Carbon loss (tC ha^{-1} yr^{-1})	Water table depth (cm)	Source	Time since draining (yr)
Murdiyarso *et al* 2010	Oil palm	MB	5.2 ± 1.1	n/a	Main text	
Hergoualc'h and Verchot 2013	Oil palm	MB	8.2 ± 2.9	66 ± 13	Table 4	
IPCC 2013	Oil palm	S/MB	12 (6.0–18)	deep	Table 2.1	>6
Couwenberg and Hooijer 2013	Oil palm and acacia	S	18	56–70	Abstract	>5
Hergoualc'h and Verchot 2013	Acacia	MB	20 ± 3.5	79 ± 7.5	Table 4	
This study	Perennial plantation	S	20 (18–22)	70	Subsidence model	>2
IPCC 2013	Short-rotation plantation	S/MB	21 (17–26)	deep	Table 2.1	>6
Page *et al* 2011	Oil palm	S	22 ± 4.6	70	Table 5	
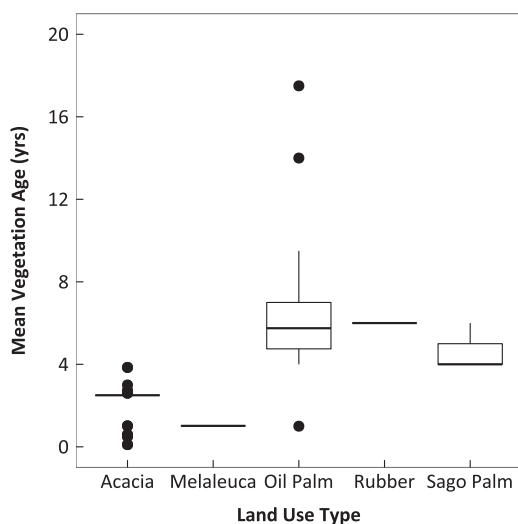
Figure 3. Mean vegetation age distribution across sites in tropical plantation land uses on peatlands in Southeast Asia. Pulpwood sites (acacia, melaleuca) are younger compared to other land uses because typical pulpwood rotations are only ~4–5 yr. Age in other land uses is skewed toward younger vegetation (e.g., oil palm planting cycles span ~25 yr, but median age is ~6 yr). Black bars represent median values; upper and lower 'hinges' are 25 and 75th percentiles; whiskers correspond to the greatest (or least) value 1.5 times the interquartile range of the hinge; and black circles are outside this range.

diffusion and transport of CH$_4$ and oxygen, altering carbon dynamics [74]. In oil palm plantations, biweekly manual fruit harvest introduces minor recurrent soil disturbance [24, 52], yet these plantations are cleared and replanted less frequently (~every 25 yr) than acacia (~every 4–5 yr). The effect of such replanting activities is unknown since studies included here did not monitor subsidence or respiration during harvest. Finally, while Hooijer *et al* [30] suggest that the depth-emission relationship may break down under

intensive fertilizer application, our findings indicate that carbon loss and soil respiration are correlated with WT depth even across fertilizer and soil disturbance regimes. Measurements from a wider range of vegetation ages, as well as better data on soil amendment and disturbance, would clarify the relative influence of and interactions between vegetation age, land use, plantation management, and WT depth on carbon loss rates.

9. Conclusions

Here, we evaluated relationships between WT depth and carbon loss from plantations on drained tropical peatlands in Southeast Asia. We found significant and positive correlations between mean WT depth and net carbon loss, heterotrophic soil respiration, and total soil respiration. Since WT depth in a single location may vary by >0.5 m in time [30, 75, 76], we stress that our models should only be applied using long-term mean WT depths (i.e., measured at frequent time intervals for $\geqslant$1 yr). While model intercept terms are not significantly different than zero, we do not suggest zero emissions at zero WT depth; models are only applicable at groundwater depths from ~20 to 110 cm.

Our analysis suggests that if plantation WTs could be raised, carbon loss rates might decline. However, the effects of WT management on radiative forcing depend partly on how WT changes affect CH$_4$ versus CO$_2$ emissions [20]. Moreover, reducing WT depths may negatively impact plantation yields [59]. Additionally, many regions experience prolonged dry seasons where maintaining shallow drainage depths may be impossible. Our assessment aligns with several recent studies [12, 23, 30, 41, 63] that indicate diverse factors besides WT depth are important to consider when quantifying and managing soil carbon losses.

While peatlands remain drained, they will continue to experience net carbon losses. Mitigating GHG emissions from tropical plantations should rely mainly on preventing further plantation expansion on to peatlands [8, 77], as well as protecting undisturbed peat swamp forests from disturbance [31, 32].

Given the divergence of total carbon loss predicted from mass balance and subsidence models, reconciling these models is a clear research priority. Our literature review suggests that improved spatio-temporal sampling will help resolve differences between models. Developing and parameterizing robust mass balance models will enable partitioning net carbon loss among different export processes, constraining the global warming contribution of peatland drainage.

Acknowledgments

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