

Article

Comparing GHG Emissions from Drained Oil Palm and Recovering Tropical Peatland Forests in Malaysia

Siti Noor Fitriah Azizan ¹, Yuji Goto ¹, Toshihiro Doi ¹, Muhammad Imran Firdaus Kamardan ², Hirofumi Hara ³ , Iain McTaggart ⁴, Takamitsu Kai ⁵  and Kosuke Noborio ^{4,*}

¹ Graduate School of Agriculture, Meiji University, Kawasaki 214-8571, Japan; azizanf@meiji.ac.jp (S.N.F.A.); fam510ego@gmail.com (Y.G.); tdoi@meiji.ac.jp (T.D.)

² Department of Chemical and Environmental Engineering, Malaysia-Japan International Institute of Technology, Universiti Teknologi Malaysia, Kuala Lumpur 54100, Malaysia; imranfirdaus@icloud.com

³ Department of Biotechnology, Graduate School of Agricultural and Life Sciences, The University of Tokyo, Tokyo 113-8657, Japan; ahara@g.ecc.u-tokyo.ac.jp

⁴ School of Agriculture, Meiji University, Kawasaki 214-8571, Japan; imctagg@meiji.ac.jp

⁵ Kurokawa Field Science Center, Meiji University, Kawasaki 215-0035, Japan; kai_takamitsu@meiji.ac.jp

* Correspondence: noboriok@meiji.ac.jp; Tel./Fax: +81-4-4934-7156

Abstract: For agricultural purposes, the drainage and deforestation of Southeast Asian peatland resulted in high greenhouse gases' (GHGs, e.g., CO₂, N₂O and CH₄) emission. A peatland regenerating initiative, by rewetting and vegetation restoration, reflects evidence of subsequent forest recovery. In this study, we compared GHG emissions from three Malaysian tropical peatland systems under the following different land-use conditions: (i) drained oil palm plantation (OP), (ii) rewetting-restored forest (RF) and (iii) undrained natural forest (NF). Biweekly temporal measurements of CO₂, CH₄ and N₂O fluxes were conducted using a closed-chamber method from July 2017 to December 2018, along with the continuous measurement of environmental variables and a one-time measurement of the soil physicochemical properties. The biweekly emission data were integrated to provide cumulative fluxes using the trapezoidal rule. Our results indicated that the changes in environmental conditions resulting from draining (OP) or rewetting historically drained peatland (RF) affected CH₄ and N₂O emissions more than CO₂ emissions. The cumulative CH₄ emission was significantly higher in the forested sites (RF and NF), which was linked to their significantly higher water table (WT) level ($p < 0.05$). Similarly, the high cumulative CO₂ emission trends at the RF and OP sites indicated that the RF rewetting-restored peatland system continued to have high decomposition rates despite having a significantly higher WT than the OP ($p < 0.05$). The highest cumulative N₂O emission at the drained-fertilized OP and rewetting-restored RF sites was linked to the available substrates for high decomposition (low C/N ratio) together with soil organic matter mineralization that provided inorganic nitrogen (N), enabling ideal conditions for microbial mediated N₂O emissions. Overall, the measured peat properties did not vary significantly among the different land uses. However, the lower C/N ratio at the OP and the RF sites indicated higher decomposition rates in the drained and historically drained peat than the undrained natural peat (NF), which was associated with high cumulative CO₂ and N₂O emissions in our study.

Keywords: drained peatland; oil palm plantation; rewetting-restored forest; GHG emissions



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1. Introduction

Greenhouse gas (GHG) emissions cause global warming, as well as contributing to the loss of carbon (C) and nitrogen (N) from soils. Three of the most important GHGs are carbon dioxide (CO₂), methane (CH₄) and nitrous oxide (N₂O), but they have different atmospheric lifetimes and radiative efficiencies. Carbon dioxide and N₂O are long-lived atmospheric gases, although the radiative efficiency of N₂O molecules is much greater than that of CO₂ [1]. Methane has only a short lifetime (12 years), but its radiative efficiency is 137-fold

greater than CO₂ when including indirect effects [1]. Peatlands play an important role in regulating GHG concentrations in the atmosphere. For example, undrained peatlands may become CO₂ sinks due to peat accumulation [2–4], while, at the same time, be CH₄ sources due to favorable methanogenesis conditions [5,6]. Previous studies have reported that GHG emissions from tropical peatlands are related to peat composition, as well as subsidence and degradation due to human activities and land-use change that affect aspects such as water table (WT) levels, vegetation cover and fire activities [7–9].

Lowland tropical peatlands are formed under a high temperature and high precipitation conditions and are estimated to cover 24.8–30 Mha in Southeast Asia [10], 46.1 Mha in South America and 18.1 Mha in central and southern Africa [11]. The peatlands in Southeast Asia are mostly distributed in Indonesia (20.6–22.5 Mha) and Malaysia (2.5–2.9 Mha) [10,11]. However, over the last 20–30 years, there have been large changes in land use on Southeast Asian peatlands [12]. For example, managed land cover (industrial plantations and smallholder dominated areas) increased from 1.7 Mha of peatland areas in 1990 to 7.8 Mha in 2015. Tropical peatland ecosystems have large carbon stocks of up to 3000 Mg C ha⁻¹, but large peat losses due to land-use change are causing emissions of GHGs of more than 20 Mg C ha⁻¹ yr⁻¹ [13]. It is estimated that the conversion of Southeast Asian peat swamp forests is contributing 16.6–27.9% of the total GHG emissions from Malaysia and Indonesia, which is equivalent to 0.440.74% of annual global emissions [14].

Over the last few decades, oil palm has become the largest source of world vegetable oil, representing 30% of global vegetable oil usage, and this has resulted in oil palm production becoming an important industry in tropical regions [15]. Oil palm cultivation on peatlands necessitates the construction of drainage ditches to lower the WT level away from the surface, with a significant impact on the function of peatlands, which is transformed from a sink to a source of GHGs (CO₂ and N₂O) [8,12,14,16]. The high CO₂ emissions occur because peatland drainage raises the soil's oxygen levels, which accelerates organic matter decomposition [14], whilst the high N₂O emissions are mainly due to nitrogen fertilization [17]. Uning et al. [18] examined the characteristics of oil palm plantations and their effects, particularly CO₂ emissions in Southeast Asia, and found that CO₂ emissions from oil palm plantations are approximately one to two times higher than emissions from other major crops. Malaysia is the second-largest oil palm producer in the world; oil palm plantations have expanded rapidly in deeper peatland areas, especially forest areas [14,19]. However, the GHG emissions caused by this expansion of oil palm plantations and overall land-use change in the peninsular region of Malaysia is still poorly recorded. Furthermore, most of the research on the impact of oil palm production has concentrated on global and national scales [14,20,21], ignoring the high degree of uncertainty in emissions existing at local levels.

The restoration of drained peatland involves rewetting to bring the water table closer to the peat surface and, thus, recreate near-natural water table conditions. The aim is to reduce aerobic peat decomposition rates as well as peat subsidence and heterotrophic respiration in order to increase peat carbon storage [22]. Rewetting can restore GHG exchange levels close to those of undrained peatland, causing reductions in CO₂ and N₂O emissions together with a rise in CH₄ emissions [22–24]. Rewetting systems have been studied at the restoration of drained peatlands in Indonesia [25,26]. The results showed that the effectiveness of restoration in reducing GHG fluxes depended on the landcover growing on the peatland, and that canal blocking strategies to reduce drainage could not eliminate the entire risk of peat subsidence. However, there have not yet been any studies conducted on the effectiveness of rewetting in Malaysian peatlands.

Various approaches have been used for measuring GHG emissions (field-based) in tropical peatlands, each with its own set of benefits and drawbacks [10]. These are primarily related to the spatial and temporal scales of measurement, e.g., subsidence monitoring that has only been used to measure CO₂ emissions [6,27,28], closed-chamber methods [6,29–31] and the eddy covariance method [32]. With the use of subsidence monitoring, it is possible to obtain a time-integrated assessment of the total carbon balance of a drained peatland

system. However, because subsidence is a slow process, it is necessary to collect subsidence data over an extended period of time (preferably a number of years, although larger numbers of measurements can compensate if measurements are for a shorter period) and to combine these data with exact measurements of peat bulk density and carbon concentration, as has been performed in several published studies [27,28,33]. However, previous studies involving closed-chamber methods have generally not been able to produce statistically robust flux values or uncertainty ranges due to inadequate numbers of replicates, and not enough close-interval measurements conducted over long enough periods of time [31,34,35]. In addition, most studies of temporal measurements have only focused on dry or wet seasons (selective time periods) [14,31,35,36] with very few studies conducting measurements for at least one full year [36–40]. Additionally, most studies of the relationships between Southeast Asian tropical peatlands and GHG emissions have been conducted in Indonesia [33,35,36,38,41,42] and east Malaysia, [7,37,43–46], with fewer studies in peninsular Malaysia peatland [14,30,34]. Hence, while the body of knowledge of GHG emissions from tropical peatlands in Malaysia has increased a little in recent years, much of this research has occurred in remote regions, and that has limited the ability to obtain detailed data. Therefore, to fill the gap in the current understanding of GHG emissions from a tropical peatland associated with conversion to oil palm plantations and then subsequent forest recovery in peninsular Malaysia, we conducted a 15-month temporal study in the northern Selangor peatlands of peninsular Malaysia. Carbon dioxide, N₂O and CH₄ fluxes were measured at biweekly intervals from the soil along with measurements of in situ environmental factors in the following three different land-uses: (1) drained matured oil palm plantation, (2) rewetting-restored forest and (3) undrained natural forest. Our study addressed the following three questions: (1) how do environmental parameters (e.g., WT, soil moisture, peat properties) affect the emission rates of CO₂, N₂O and CH₄? (2) How does land cover (oil palm plantation or forest, and forest type) affect GHG emissions? (3) How important is the relative contribution of the annual CO₂, N₂O and CH₄ fluxes in our study sites to global warming compared with the equivalent annual emissions in other land-use studies on tropical peatlands? We hypothesized that drained tropical peatland, e.g., oil palm plantation, would emit significantly more GHGs than a natural peatland system. We expected that implementing hydrological restoration on previously drained peatland may reduce GHGs' emissions, e.g., CO₂ and N₂O.

2. Materials and Methods

2.1. Study Sites

North Selangor Peat Swamp Forest (NSPSF) consists of two natural reserves, which are Raja Musa and Sungai Karang Peat Swamp Forest Reserve (PSFR). This study was carried out in Raja Musa PSFR. There were the following three different sites: (1) a drained 10–12-year-old matured oil palm plantation (OP) where the oil palms were part of the first crop rotation (3°25'29.92" N 101°20'10.47" E), (2) a 6-year-old recovering (by rewetting and replanting) peat swamp forest (RF) that had been illegally cultivated with oil palm by smallholders for 10 years before being recovered and managed as restored peatland by the Selangor State Forestry Department in 2012 (3°25'35.99" N 101°20'08.6" E) and (3) an approximately 45-year-old regenerating undrained natural peat swamp forest (NF) that had been logged for timber in the 1980s (3°26'20.29" N 101°20'02.28" E) (Figure 1). Raja Musa PSFR is part of the North Selangor Peat Swamp Forest (NSPSF), covering 23,486 hectares of the 81,304 hectares in NSPSF. Part of the Raja Musa PSFR was burned in 2012, but none of the three selected sampling areas were affected by those fires (Selangor Forestry Department, personal communication). However, both the RF and OP sites are assumed to have been subject to burning as part of the land clearing process when they were originally converted from natural peatland to agricultural land [47]. The biomass of the RF is a mix of oil palm trees and mixed swamp forest species, including replanted Tenggek burung (*Euodia redleyi*). The NF is an intact peatland, which contains large areas of forest cover and high-water tables. The dominant tree species within the NF site are

Kempas (*Koombassia malaccensis*), Kedondong (*Santiria* spp.) and Kelat (*Syzgium* spp.). More information about NSPSF is provided by Tonks et al. [20], Dhandapani et al. [31] and Cooper et al. [14]. In general, the mean annual rainfall in NSPSF varies from 1359 to 2480 mm, peaking in October–November with the least amount of rain occurring in May–September [48,49]. In this study, rainfall data from July 2017 to December 2018 were obtained from the 3 weather stations (Batang Berjuntai, Sungai Karang and Sungai Tenggi Kiri) to account for the local variability of rainfall in the Raja Musa PSFR area. The rainfall data for Batang Berjuntai and Sungai Karang were provided by World Weather Online (accessed on 13 October 2021) [50] and the data for Sungai Tenggi Kiri were provided by AccuWeather.com (accessed on 13 October 2021) [51]. Mean precipitation from July 2017 until December 2018 ranged from 25.05 to 415.69 mm. The dry periods were recorded in July 2017 and July–August 2018 (range 34–49 mm/month), and the highest rainfall was recorded at all three weather stations between October and December 2018 with an average precipitation over the three months of 374.66 ± 98.48 mm, in what was considered to be the wet season. The Rasa Musa area received very little rain at the end of 2017, and there was no significant change in rainfall between August 2017 and June 2018.



Figure 1. Study sites at Raja Musa Peat Swamp Forest Reserve. OP: oil palm plantation, RF: rewetting-restored forest and NF: natural forest.

2.2. Sampling Strategy

A data logger (Em50, Decagon Devices, Pullman, WA, USA) was installed at each sampling site prior to gas sampling to measure the following environmental variables at 10 min intervals: Water Table level (WT), atmospheric temperature, soil moisture and soil temperature. These variables were measured from July 2017 to September 2018 at the OP and NF sites, and from July 2017 to October 2018 at the RF site. Greenhouse gases were sampled using the static closed-chamber method [52] every two weeks, between 11:00 and 14:00 h on each sampling day, from July 2017 to December 2018 at the OP and RF sites and from July 2017 to September 2018 at the NF site because, after that date, the sampling site became inaccessible due to flooding. The static closed-chamber method has a clear advantage over other methods in that it is the only operational method that can obtain temporal measurements of trace gas fluxes at a large number of plots spread across a large area, as well as yielding information on the short-distance spatial variability of trace gas fluxes [53]. The chambers in the current study consisted of white-painted galvanized-steel cylinders with a height of 25 cm and an inner diameter of 20 cm, that were open at one end (bottom end) and sealed at the other (top) end. A small fan was equipped inside the chamber (at the top) to mix the air. Three static closed chambers were set up at each site on each sampling event. In order to minimize any influence of autotrophic respiration on the chamber gas fluxes, all the chambers were at least 2 m from the nearest big tree in the RF and NF sites and at least 3 m away from any mature oil palm trees in the OP site [54]. For the gas flux measurement, the chambers were inserted into an un-vegetated area of peat

to an approximately 2-centimeter depth to provide a gas seal around the chamber. Gas sampling did not start until 3 min after chamber installation to allow for gas stabilization. The 30-milliliter gas sample was collected using a syringe to withdraw gas from a sampling port fitted to the top of the chamber and transferred into evacuated vial. Gas collections were taken at 0, 10 and 20 min. The vials were kept in an ice box during transportation back to the local laboratory in Malaysia and then stored at 4 °C until transport to Japan for gas analysis. All samples were transported to Japan by the end of December 2018.

2.3. Peat Sampling and Analysis

Three replicate 1-kilogram composite peat samples were taken from the top 10 cm soil depth at each site at the start of the gas sampling period. After sampling, the peat samples were oven-dried at 60 °C until they were a constant weight, and then bulk density was determined from the dry mass and the volume of the ring sampler. Particle density was determined with the picnometric method using a three-phase meter [55,56]. Volumetric soil water content was measured using the gravimetric method as the mass of water per mass of dried soil and is expressed as a ratio [57]. The soil pH was analyzed using a pH meter (Model: LAQUA F-72, Horiba Scientific, Kyoto, Japan). The total carbon (TC) content was analyzed with a TOC analyzer (Model: SSM-5000A, Shimadzu, Kyoto, Japan). The following chemical properties were measured in peat extracts that were prepared by shaking a soil-water suspension (1:20, *w/v*) at 100 rpm for 1 h: ammonium-nitrogen (NH_4^+ N), nitrate-nitrogen (NO_3^- N), total nitrogen (TN), total phosphorus (TP) and total potassium (TK). The NH_4^+ N and NO_3^- N contents were analyzed with 1 M KCl and the brucine method [58], respectively. The TN, TP and TK contents were analyzed by digesting soils in a Kjeldahl Therm digestion unit (Gerhardt, Königswinter, Germany) with H_2SO_4 and H_2O_2 [55]. Water-filled pore space (WFPS) values were calculated using the following equation: $\text{WFPS} = [\text{SWC}/(1 - (\text{BD}/\text{PD}))] \times 100$, where SWC is the volumetric soil water content (vol. %), BD is the soil bulk density (g cm^{-3}) and PD is the particle density (g cm^{-3}) [59].

2.4. Greenhouse Gases Measurement

All gas samples were analyzed in April–June 2019 using gas chromatography (Agilent Technologies 6890, California, SC, USA) with a flame ionization detector (FID) for CH_4 and CO_2 and an electron capture detector (ECD) for N_2O . The gas chromatograph measurements in ppm were then converted to gas concentration values of mg m^{-3} using the ideal derivative gas law equation,

$$W = MVP/RT \quad (1)$$

This equation was derived from $PV = nRT$, where P is pressure (1 atm), V is volume (ppm), n is the number of moles of gas, R is the universal gas constant ($0.082 \text{ atm L mol}^{-1}\text{K}^{-1}$) and T is the temperature in degrees kelvin (K). Hence, $n = W/M$, where W is the weight of the gas (mg m^{-3}), and M is the molar mass of the gas (g mol^{-1}). Therefore, the gas emission was then calculated using the following linear regression equation [60]:

$$F = \Delta c/\Delta t \times h \quad (2)$$

where F is the gas flux ($\text{mg m}^{-2} \text{ h}^{-1}$), $\Delta c/\Delta t$ is the change in gas concentration inside the sampling period and h is the height inside the chamber above the soil surface (m). The cumulative emissions ($F_{cum,t}$) were calculated from the values at each sampling time. First, the arithmetic mean of the gas flux, ($\text{mg m}^{-2} \text{ h}^{-1}$) at each site (3 replicate chambers) was calculated for each sampling time. Then, the fluxes at each sampling date were interpolated and integrated over time using the trapezoidal rule. The area of each trapezoid formed between sampling time data points were calculated and summed to give the cumulative flux in mg m^{-2} [60].

Global Warming Potential (GWP) is an indicator of the radiative forcing of a greenhouse gas over a chosen time horizon, relative to CO_2 . To compare the overall GWP load

of the three GHGs, annualized emissions of CH₄ and N₂O were converted to CO₂ equivalents (CO₂-eq) over a 20-year horizon because this time period was considered to be the most appropriate for evaluating the impacts of land-use change that typically occur over 20–30-year time periods in tropical regions [14,40]. Annualized emissions were calculated using the cumulative emissions multiplied by a constant of 365/total days of cumulative emission. The GWP value for CH₄ (20-year horizon) is 86 and for N₂O it is 268, based on the IPCC Fifth Assessment Report excluding climate-carbon feedback [61]. We used the calculated CO₂-eq annual emissions to compare against the annual emissions derived from other land-use studies in tropical peatlands.

2.5. Data and Statistical Analysis

All statistical analyses were carried out using EXCEL 2018 (Microsoft Corporation, Redmond, WA, USA). Pearson's correlation was used to determine the correlations between GHG emissions and hydrology parameters variables. Then, a one-way ANOVA with post hoc test was used to compare GHG emissions and environmental parameters at the different sites. Calculated cumulative gases flux values were used to compare the GHG emissions between sites, while mean values were used for environmental parameters. The statistical significance level was set at $p = 0.05$.

3. Results

3.1. Peat Characteristics

The high soil organic C and N at the three sites is a typical characteristic of tropical peatland (Table 1). The OP soil had a higher bulk density and particle density than the two forested sites, which resulted in a lower porosity. The higher surface organic matter content in the two forested sites would have reduced the particle density and bulk density compared to the OP. There was no significant difference in the WFPS between sites, although the NF was slightly higher than the OP and RF sites. The C/N ratio of the NF was much higher than at either the RF or OP sites. The NH₄⁺-N content was notably higher in the RF than in either the OP or the NF, while the NO₃⁻-N content was much lower than the NH₄⁺-N content at all three sites, with the NO₃⁻-N content at the OP being slightly higher than at the other two sites. However, overall, there were no statistically significant differences in the measured peat properties between the sites in this study.

Table 1. Physicochemical properties of peat (mean ± STD) at the drained oil palm plantation (OP), rewetting-restored forest (RF) and undrained natural forest (NF).

Properties	OP	RF	NF
Bulk density (gcm ⁻³)	0.21 ± 0.00	0.18 ± 0.02	0.17 ± 0.00
Particle density (gcm ⁻³)	1.53 ± 0.007	1.43 ± 0.010	1.43 ± 0.005
WFPS (%)	74.4651	72.4581	77.6919
Ph	3.17 ± 0.12	3.33 ± 0.15	3.50 ± 0.26
WT (cm)	−68.94 ± 244	−18.38 ± 8.46	4.92 ± 17.37
Soil surface temp. (°C)	27.55 ± 04	27.51 ± 0.01	26.02 ± 0.95
Air temp. (°C)	26.86 ± 3.98	26.81 ± 4.14	26.09 ± 2.94
Total C (%)	14.36 ± 22.05	17.821 ± 28.28	16.678 ± 27.10
Total N (%)	1.53 ± 1.65	0.40 ± 0.44	0.19 ± 0.03
C/N ratio	16.46 ± 19.54	25.06 ± 26.19	76.37 ± 121.71
NO ₃ ⁻ -N (mg N Kg ⁻¹)	3.00 ± 2.66	1.00 ± 0.01	0.00 ± 0.00
NH ₄ ⁺ -N (mg N Kg ⁻¹)	21.67 ± 2.08	66.67 ± 28.31	18.33 ± 23.97

WFPS: water-filled pore space, WT: water table level, C: carbon, N: Nitrogen, NO₃⁻-N: nitrate-nitrogen, NH₄⁺-N: ammonium-nitrogen. Negative value for WT represents below the surface.

3.2. Variation in Environmental Variables

The average air temperatures during the study at the OP, RF and NF sites were 26.86 ± 3.98 °C, 26.81 ± 4.14 °C and 26.09 ± 2.94 °C, respectively (Figure 2a). The soil surface temperature at all the sites was significantly different from the air temperature

($p < 0.05$), but there was a strong correlation between the soil surface and air temperatures at the OP site, $r^2 = 0.88$, $p < 0.05$. Additionally, the soil surface temperature at the NF site was slightly lower than at the RF and OP sites (Figure 2b). Soil moisture was significantly lower at the OP site compared to the RF and NF forested sites (Figure 2c), and also the OP had a much lower WT level than either the NF or the RF (Figure 2d; $p < 0.05$). There was a strong correlation between the soil moisture and WT level at the RF ($r^2 = 0.77$, $p < 0.05$), Figure 3, and the various rising pulses in the WT levels at all three sites were potentially due to rainfall events (Supplementary Figure S1).

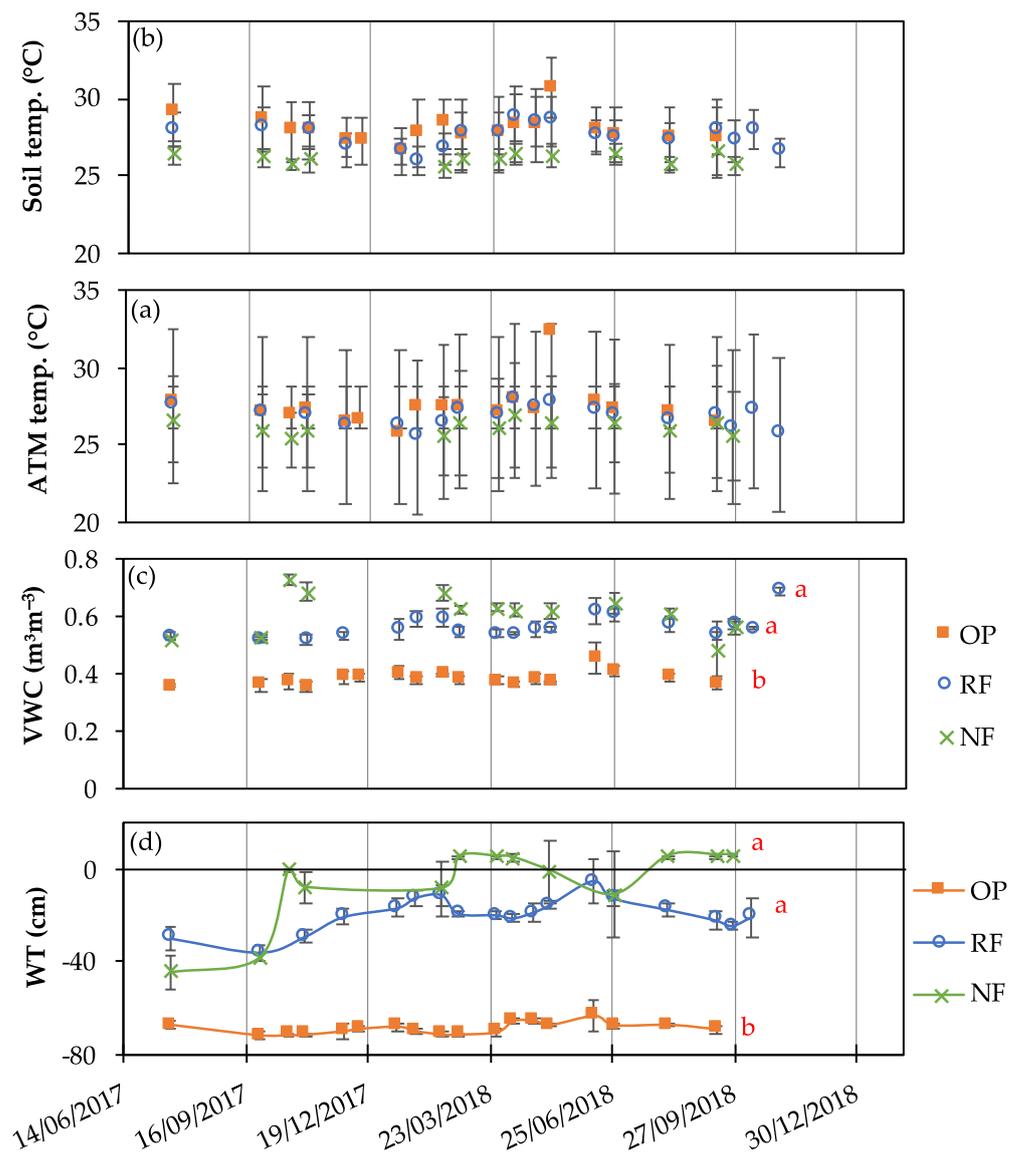


Figure 2. Environmental parameters at the drained oil palm plantation (OP), rewetting-restored forest (RF) and natural peat swamp forest (NF) sites in Raja Musa, Malaysia. (a) atmospheric temperature, (b) soil temperature, (c) soil moisture and (d) water table level. ATM: atmospheric. Error bars represent standard error of the mean. Note: different letters in red represent significantly different, $\alpha = 0.05$.

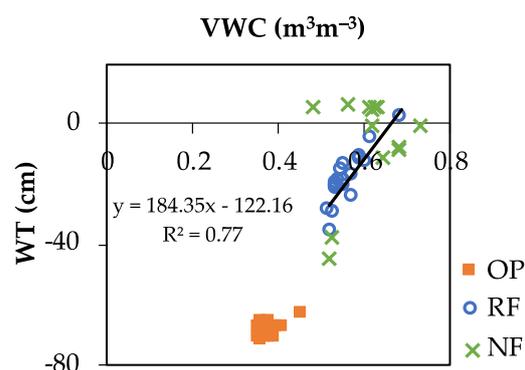


Figure 3. Relationship between water table level and soil moisture at at the oil palm plantation (OP), recovery peat swamp forest (RF) and natural peat swamp forest (NF) sites in Raja Musa, Malaysia.

3.3. Variation in GHGs Emission

The average hourly CO_2 , N_2O and CH_4 fluxes over 15 months at the OP, RF and NF sites are shown in Figure 4a–c, respectively. The box and whisker plots show the 25, 50 (median) and 75% quartiles for the emission range of each greenhouse gas. There were notable differences between the sites for the flux ranges of N_2O and CH_4 than there were for CO_2 (Figure 4). For N_2O , the flux range at the OP site was much greater than at the RF or NF sites (Figure 4b), whereas for CH_4 , the flux range at the NF site was much greater than at either the RF or OP sites (Figure 4c).

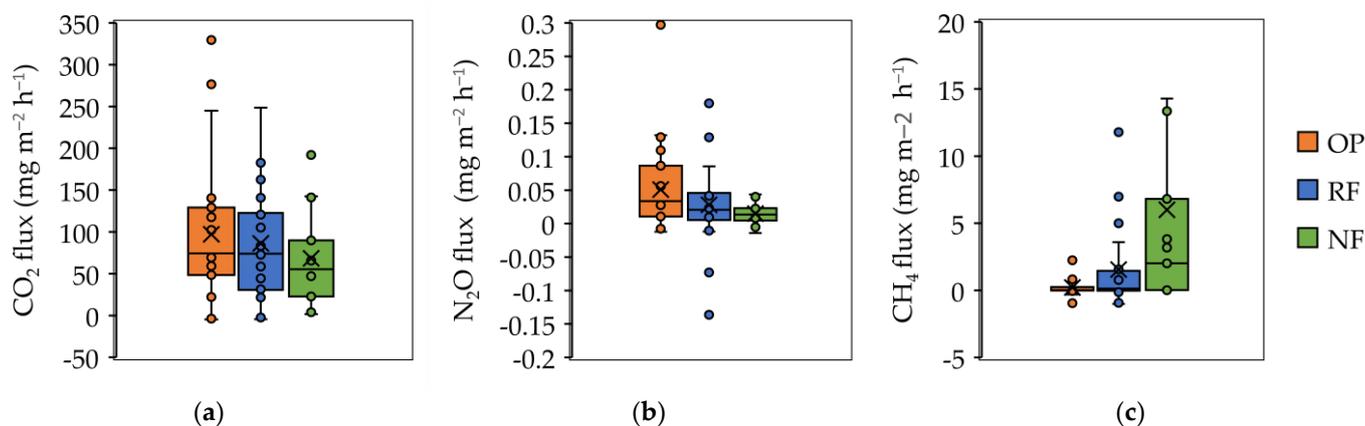


Figure 4. Box and whiskers plot showing 25%, median and 75% quartiles. \bar{x} is the mean value, 95% confidence interval (whiskers) and outlier emission values at the drained oil palm plantation (OP), rewetting-restored forest (RF) and undrained forest (NF) sites in Raja Musa, Malaysia. (a) CO_2 , (b) N_2O and (c) CH_4 .

There was little difference between the CO_2 flux ranges in the disturbed RF and OP peatlands (Figure 4a). The average hourly and cumulative CO_2 emissions from the NF site were lower than from the disturbed OP and RF sites, but the differences were not significant (Figures 4a and 5c). The cumulative CO_2 flux trend for the OP and RF sites was very similar (Figure 5a).

Nitrous oxide fluxes at the disturbed OP and RF sites were more variable than at the undisturbed NF site (Figure 4b). Additionally, the cumulative N_2O emissions from the OP and RF sites were significantly higher than at the NF site, but there was little difference between the OP and RF sites (Figure 5b). The most rapid increase in cumulative N_2O occurred between August and September 2018 in the OP and from June to September 2018 in the RF (Figure 5b). At the RF site, there were measurements of N_2O uptake between late October and early November 2018, resulting in a reduction in cumulative N_2O emissions during this period (Figure 5b).

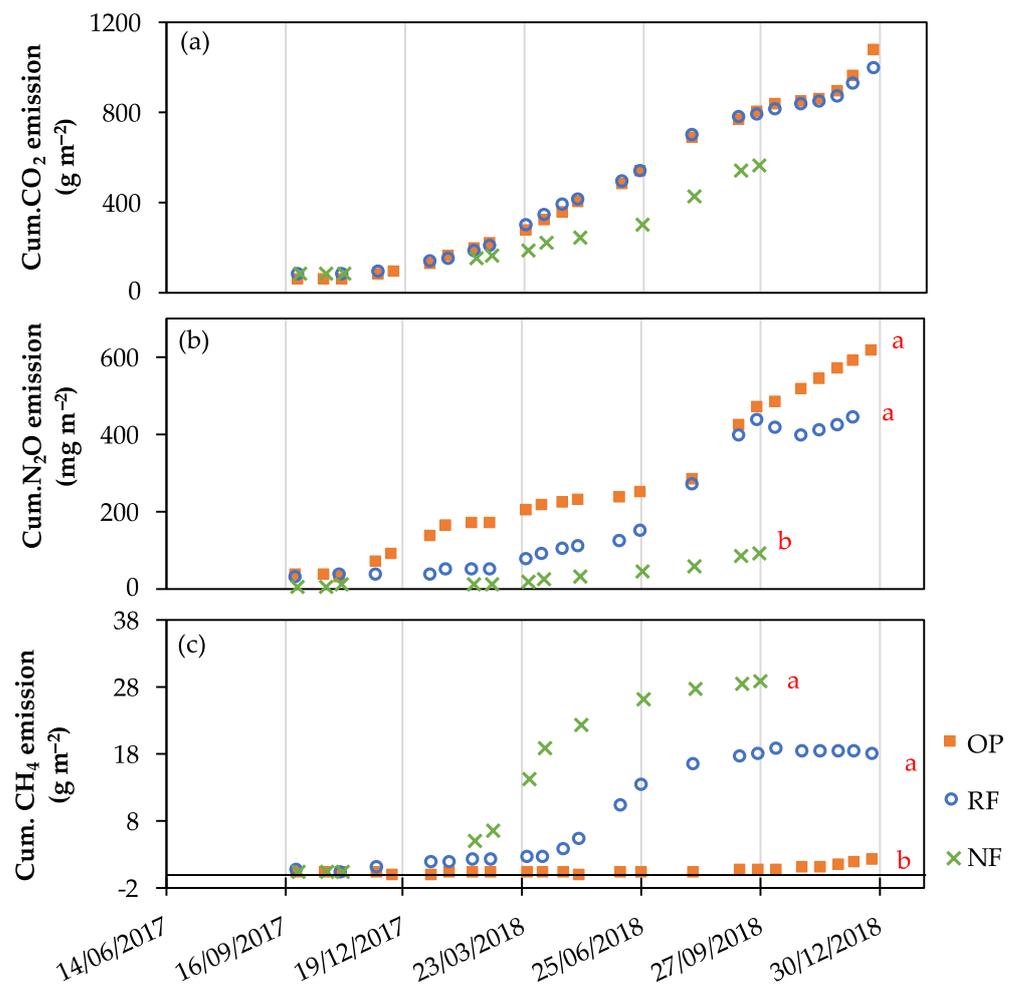


Figure 5. Cumulative emission of (a) CO₂, (b) N₂O and (c) CH₄ at the oil palm plantation (OP), recovery peat swamp forest (RF) and natural peat swamp forest (NF) sites in Raja Musa, Malaysia. Note: different letters represent significantly different, $\alpha = 0.05$.

The average hourly CH₄ fluxes from the NF, RF and OP sites were $5.99 \pm 9.85 \text{ mg m}^{-2}\text{hr}^{-1}$, $1.55 \pm 3.94 \text{ mg m}^{-2}\text{hr}^{-1}$ and $0.21 \pm 0.59 \text{ mg m}^{-2}\text{hr}^{-1}$, respectively (Figure 4c). Cumulative emissions from the forested NF and RF sites were significantly higher than from the OP ($p < 0.05$), approximately 18 and 11 times higher, respectively (Figure 5c). There was no significant difference in the cumulative CH₄ emissions between the NF and RF sites. The highest CH₄ fluxes at the NF and RF sites occurred between late March and early June 2018, when the WT levels were at their highest (Figures 2d and 5c). Methane uptake was observed for over 50% of the measurements at the OP site, and there was very little cumulative CH₄ emission from the OP site (Figure 5c).

3.4. Global Warming Potential (GWP)

In this study, CH₄ and N₂O annualized emissions were converted into CO₂ equivalents to estimate the 20-year GWP of total emissions from the different peatland sites and to determine the relative contributions of each greenhouse gas. The total 20-year GWP annualized GHG emissions were highest at the undrained peatland (NF), which were 2.8 magnitudes higher than the currently drained peatland (OP). Additionally, the 20-year GWP annualized GHG emissions were notably higher at the forested sites (NF: $2616.42 \text{ g CO}_2 \text{ eq. m}^{-2}\text{yr}^{-1}$ and RF: $1808.91 \text{ g CO}_2 \text{ eq. m}^{-2}\text{yr}^{-1}$, respectively) due mainly to higher CH₄ emissions. The CH₄ emissions at these two sites accounted for 81 and 58%, respectively, of the 20-year GWP emissions in the forested peatlands. The disturbed-peatlands, consisting of the currently drained OP site and the recently rewetted RF site,

had higher annualized CO₂ emissions (680.1–726.9 g m⁻² yr⁻¹) than the NF undrained peatland system (Table 2). Therefore, at the OP site, the major contribution to annualized GWP emissions was CO₂, accounting for 78% of the total 20-year GWP. On the other hand, the highest contribution of N₂O emissions to the cumulative GWP was at the OP site (12% of the total GWP at that site; Table 3), but it only contributed 4 and 1% of the total 20-year GWP at the forested RF and NF sites, respectively (Table 3).

Table 2. Annualized emission of CO₂, N₂O and CH₄ (g m⁻² yr⁻¹) at the oil palm plantation (OP), recovery peat swamp forest (RF) and natural peat swamp forest (NF) sites in Raja Musa, Malaysia.

Sites	Annualized Emission of CO ₂ , N ₂ O and CH ₄ (g m ⁻² yr ⁻¹)		
	[Mean Value (mg m ⁻² h ⁻¹) ± stv]		
	CO ₂	N ₂ O	CH ₄
OP	726.99 [97.05 ± 84.42]	0.42 [0.05 ± 0.06]	1.09 [0.21 ± 0.59]
RF	680.06 [86.23 ± 66.26]	0.29 [0.03 ± 0.06]	12.19 [1.55 ± 3.04]
NF	481.93 [68.33 ± 59.31]	0.08 [0.01 ± 0.02]	24.57 [5.99 ± 9.85]

stv: standard deviation.

Table 3. 20-year GWP of Annualized emissions (g CO₂ eq.m⁻² yr⁻¹) of peatland sites in the current study and published results from other studies.

Sites	20-Years GWP (g CO ₂ eq.m ⁻² yr ⁻¹) [% of Total GWP]			
	CO ₂	N ₂ O	CH ₄	Total
Drained, uncleared forest (Pen. Malaysia) [14]	5632 ± 13.15 [54]	5562 ± 21.47 [53]	52 ± 0.23 [0.5]	10,486 ± 22.88
6 months OP (Pen. Malaysia) [14]	8137 ± 9.54 [32]	18,242 ± 78.03 [71]	268 ± 1.20 [1]	25,853 ± 75.43
OP (this study)	726.98 [78]	112.91 [12]	93.62 [10]	933.52
10–15 years old OP (Pen. Malaysia) [14]	5441 ± 13.25 [56]	3434 ± 15.78 [35]	767 ± 3.43 [8]	9736 ± 21.68
Matured OP (Kal. Indonesia) [40]	3040 ± 8.5 [96]	130 ± 1.0 [4]	0.0 ± 0.0 [0]	3170 ± 8.6
RF (this study)	680.06 [38]	80.15 [4]	1048.69 [58]	1808.91
30-yo secondary forest (Kal, Indonesia) [40]	520 ± 4.4 [51]	340 ± 0.8 [33]	160 ± 0.5 [16]	1020 ± 4.5
NF (this study)	481.93 [18]	21.10 [0.8]	2113.39 [81]	2616.42
Primary forest (Kal, Indonesia) [14]	−50 ± 3.9 [−51]	40 ± 0.2 [42]	100 ± 0.2 [109]	90 ± 3.9

-yo: years old, Pen. Malaysia: peninsular Malaysia, Kal. Indonesia: Kalimantan Indonesia, OP: drained oil palm plantation, RF: rewetting-restored forest and NF: undrained natural forest. The sites were arranged to demonstrate how agricultural conversion affected recovering peat swamp forest system, e.g., (i) drained forest and early oil palm conversion, (ii) matured oil palm plantation, (iii) recovery forest and (iv).

4. Discussion

4.1. Peat and Environmental Characteristics

The bulk density of the OP surface peat (0.21 g cm⁻³) in this study was still within the range of the FAO range for tropical peat (0.05 and 0.5 g cm⁻³). It was also comparable to the values found in previous studies in the same region for first generation matured oil palm plantations (0.13–0.32 g cm⁻³), but lower than the values for second generation matured oil palm plantations (0.43 g cm⁻³) [20]. The lower bulk densities at the RF and NF forest sites in our study were also comparable to those found in Southeast Asian tropical peatlands, which range from 0.02 to 0.21 g cm⁻³ [10,14,62,63]. It has been shown that the agricultural drainage-based conversion of peatland enhances various processes including peat shrinkage, compaction and solidification, which result in an increase in peat bulk density [28,64–66]. Thus, the higher bulk density at the OP site could have been caused by particulate organic matter being removed from the surface peat layer [67] as well as compaction caused by the aerobic decomposition of the deeper layers of unsaturated peat [68] and solidification by the mass of the oil palm tree stand.

Studies on disturbed Selangor peatlands have shown that the decomposition of mature oil palm surface peats is accelerated by lower WT levels and altered litter inputs [20,54],

resulting in a gradual decrease in peat moisture content, which could be linked to peat subsidence after drainage [69]. In our study, there were similar reductions in the WT levels and soil moisture in the OP compared with the RF and the NF (Figure 2c,d) [20,54]. Additionally, the significantly lower C/N ratio in the peat surface layer at the drained OP site compared to the saturated NF site (Table 1) suggests that the loss of canopy, resulting in the greater exposure of surface peat to the air, can increase oxidative decay that leads to increased C loss. In addition, fertilizer inputs in palm oil plantations will stimulate rapid organic matter decomposition [20,69,70]. Conversely, the enhanced decomposition that leads to the low C/N ratio at the rewetting RF site that had a significantly higher WT level (Table 1, Figure 2d) may have been due to a combination of (i) the high availability of fine root and leaf litter deposited under the moderately moist and oxic conditions of the unsaturated peat layer near the surface [28] and (ii) the increased aerobic microbial decay in the root zone due to greater aeration associated with the large tree root biomass [71,72] and an open pore structure provided by the fibrous wood input. The higher C/N ratio in the saturated NF surface layers (Table 1, Figure 2d) suggests poorly decomposed peat, which is a typical characteristic of tropical peatlands [4,10,31,73].

Total C and organic matter in tropical peatland are generally positively correlated with peat acidity. In a tropical swamp forest, draining the peat increased the peat pH, leading to a decline of organic matter [74]. In contrast, the lower pH value (acidic) at the OP site could have affected the organic matter content. Hence, even though the atmospheric temperature was very similar for all three sites, the soil temperature at the OP and RF sites was one degree higher than at the NF site, probably due to their lower canopy cover. Furthermore, the high NH_4^+ -N at the RF compared to the OP and the NF suggests very high N mobilization from strongly decomposed peat in the unsaturated layer near the surface, where a greater abundance of fresh litter inputs would have been deposited [6].

4.2. Carbon Dioxide (CO_2) Emissions

The carbon dioxide fluxes at the drained OP site ($22\text{--}330 \text{ mg CO}_2 \text{ m}^{-2} \text{ h}^{-1}$) in this study are comparable to the heterotrophic fluxes previously reported for oil palm monoculture on tropical peatlands (6-year-old plantation: $114\text{--}353 \text{ mg CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ [75], 15-year-old plantation: $381 \text{ mg CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ [75], matured plantations: $46\text{--}335 \text{ mg CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ [7], smallholders' plantation, 17–18 years old: $620 \text{ mg CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ [54]). The annualized CO_2 emissions at the OP site ($726.99 \text{ g m}^{-2} \text{ yr}^{-1}$; Table 3) were seven times lower than the recent peat-oxidation emission values for tropical peatland set by the IPCC [1,76], which recommended default values of $5100 \text{ g CO}_2 \text{ m}^{-2} \text{ yr}^{-1}$ for smallholder systems and $5500 \text{ g CO}_2 \text{ m}^{-2} \text{ yr}^{-1}$ for commercial plantations (oil palm, industrial timber). The measured rates of CO_2 emissions from drained oil palm plantations have varied widely, especially between large-scale plantations and smallholder mosaic landscapes, where there can be big differences in years since drainage, peat maturity [42] and land-use practices [10,33,77,78]. Moreover, the annualized CO_2 emissions at the OP site were at least three- to four-times lower than for other agriculture practices on Southeast Asian peatlands, except for a study on sago plantations: $520 \pm 5.1 \text{ g CO}_2 \text{ m}^{-2} \text{ yr}^{-1}$ [79], e.g., croplands and shrublands: 4100 ± 6.7 , rice fields: 2560 ± 11.5 , *Acasia crassicaarpa* plantation: $7180 \pm 12.7 \text{ g CO}_2 \text{ m}^{-2} \text{ yr}^{-1}$. Wakhid et al. [38] also reported $3293 \pm 1039 \text{ g CO}_2 \text{ m}^{-2} \text{ yr}^{-1}$ emissions for eight-year-old rubber plantations (Supplementary Table S1). However, these published CO_2 emissions on drained agricultural systems in tropical peatland range widely and may reflect differences in local situations [79], such as (1) the type of plantation, e.g., high CO_2 emissions in *Acasia crassicaarpa* plantations, possibly because of *Acasia* trees that may alter soil characteristics by raising organic carbon and total nitrogen levels, resulting in increased microbial biomass and basal respiration, and (2) the variation in sampling methods, e.g., chamber-based estimation [38], transect in separation root from peat-based respiration [75,80], subsidence measurement [66], or bias due to impacts of nearby crops and canal edge effects [30,34,54,81]. For example, while Wakhid et al. [38] found no significant difference in the CO_2 emissions from a chamber positioned near or far away from rows

of trees in a rubber plantation, Dhandapani et al. [54] found significant differences in CO₂ emissions in an oil palm plantation depending on the chamber position (cleared-dead wood (highest emission) > understory fern > cleared-open area that was free of any understory vegetation > near to the canal) with consistently higher emissions from chambers within 3 m of the nearest palm tree due to the contribution of roots. Hence, Matysek et al. [34] found no root contribution to CO₂ emission for the measurements that are 3 m away from an oil palm tree. Studies based on peat subsidence measurements have estimated CO₂ emissions of 10,000 g CO₂ m⁻² yr⁻¹ from peatland at the end of a completed oil palm cultivation cycle (>25 years) [28] and 8000 g CO₂ m⁻² yr⁻¹ from peatland during early conversion (31–46 months old) [28].

Carbon dioxide emissions have been linked to drained peatland, where substantial and rapid responses to changes in the WT level have been notable [5,14]. In this study, the CO₂ emissions at the OP site where the WT level was −0.8 to −0.7 m below the ground was caused mainly by the enhancement of peat decomposition, which was evidenced by the lower C/N ratio compared to the undisturbed NF site (Table 1) [82]. Thus, the slightly higher bulk density of the drained OP compared to the undisturbed forested site suggests the progressive decomposition of organic matter [42]. The observed high cumulative CO₂ emissions in the drained OP site (Figure 5a) was considered to be due to the increased oxygen availability, which lead to aeration and greater aerobic respiration [83]. The effect of peat drainage indicates that CO₂ emissions are positively correlated with lowering the WT level [5,14] where heterotrophic respiration dominates (up to 82%) the total respiration from drained peatlands [29,77]. Marwanto et al. [42], in their study, found that CO₂ production in mature oil palm plantations is derived from 50 to 10 cm below the surface (subsoil), where the active decomposition layer of peat occurs, and is more pronounced by the moisture condition above the WT level.

Rewetting is a long-term strategy for restoring a peatland's C sink function, requiring both a high WT level and revegetation [22]. The high observed cumulative CO₂ emissions at the RF site are potentially due to a combination of root respiration from trees and high heterotrophic respiration in the subsoil in response to the high litter deposition that increased the supply of the labile C fraction, which is crucial for decomposition, specifically in peat enriched with recalcitrant substrates [6,84]. In addition, the high intensity of understory vegetation at the RF site may have potentially contributed to increasing the input of the labile C fraction by either rhizodeposition or priming effects that increased the decomposition of soil organic matter [85,86]. Murdiyarso et al. [36] reported equal partitions of heterotrophic (48%) and autotrophic (52%) respiration rates in restored peat swamp forests in Kalimantan, Indonesia.

Hydrological reestablishment brings the WT level closer to the peat surface, which allowed the RF site to return to near natural WT level conditions (Figure 2d), thus, reducing the aerobic peat decomposition rates. The WT level at the RF site (varying between −19 cm below and 2 cm above ground) was within the range indicated by Wösten et al. [87], where it should be retained between 40 cm below and 100 cm above the peat surface for the prevention of fire, peat subsidence and continuous CO₂ emissions. Murdiyarso et al. [25] found that lowering the WT level to as deep as −40 cm in both restored shrub and secondary forest increased CO₂ emission rates by 40–75%. The CO₂ emissions at the RF in our study were highly influenced by the understory vegetation with an increasing leaf litter and labile carbon addition on the ground surface [20] rather than hydrological recovery, e.g., the closed emission of cumulative CO₂ was recorded at the RF and the OP, yet they had significantly different WT levels. Some studies have suggested that vegetation is the dominant control of CO₂ emissions in the early stages of rewetting, e.g., depending on the vegetation species (*sphagnum*-dominated > shrub-dominated peatlands) and amount of vegetation cover that colonize the peatlands [88,89]. Another study found significant vegetation communities in mitigation of CO₂ emission through soil C sequestration [90].

The CO₂ fluxes recorded at the RF (21.63–248.6 mg m⁻² h⁻¹) were relatively lower compared to other rewetting tropical peatlands (shrubs, 599 mg m⁻² h⁻¹ [25]; secondary

peat swamp forests, $490 \text{ mg m}^{-2} \text{ h}^{-1}$ [25]). Additionally, the annualized CO_2 emissions at the RF site was lower compared to a study conducted by Murdiyarso et al. [25] (Supplementary Table S2). Considered at an early stage of rewetting-restored peatland (6 years), rewetting does not affect CO_2 emissions, where the cumulative CO_2 emissions at the RF were not different from the OP (Figure 5a). Furthermore, a study by Bartolucci et al. [91] found significantly high CO_2 emissions at newly restored rather than old-restored sites of a retired cranberry farm restored to a wetland ecosystem in New England, which indicates that restored systems may transition from CO_2 sources to CO_2 sinks over a larger time scale.

The low canopy coverage at the OP (monoculture palm trees) and the RF (dominated with shorter trees and shrubs) reduced shading and increased the peat surface temperatures compared to the NF (Figure 2b). This increase in surface temperature could have contributed to the higher CO_2 emissions by increasing the aerobic decomposition rate [6,92]. Other studies have also noted higher CO_2 emissions at higher soil temperatures. For example, Murdiyarso et al. [25] noted that a significantly higher soil temperature in a secondary peat swamp with shrubs ($31.2 \text{ }^\circ\text{C}$) had increased the CO_2 emissions compared with restored secondary peat swamp forests ($26.3 \text{ }^\circ\text{C}$). Additionally, other studies have reported increased CO_2 production from flooded, low oxygen peats when the temperature increased [29,93]. However, some studies have also shown that when the soil becomes dry, the respiration response to rising temperature weakens, leading to lower respiration rates [94]. Additionally, Sjögersten et al. [93] found that heterotrophic microbial communities in tropical peat respond weakly to warming, mainly due to adaptation to high peat temperatures.

The CO_2 fluxes from the undrained NF site in this study (range $1.57\text{--}192 \text{ mg CO}_2 \text{ m}^{-2} \text{ h}^{-1}$, Figure 4a) were notably lower than those in previous studies at the North Selangor swamp forest (closed to $1000 \text{ mg CO}_2 \text{ m}^{-2} \text{ h}^{-1}$) and the Terengganu swamp forest (closed to $500 \text{ mg CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ in peninsular Malaysia [31]. The annualized CO_2 emissions at the NF were comparable to other reported emissions in the same region [14], but it were lower than the data recorded for Southeast Asian peat Swamp Forest [4,36,43] and higher than values for undrained natural bog ($180 \text{ g m}^{-2} \text{ yr}^{-1}$ [95]). The CO_2 emissions from the natural NF forest in this study were not directly correlated with the measured environmental factors, which is inconsistent with previous studies in tropical peatlands, where the WT level [5,31,96], decomposability and quantity of the organic material available for decomposition (litter, peat and root exudates) and peat nutrient status [84,92] have all been shown to be correlated with CO_2 emissions. The unexpectedly low organic matter decay rates (high C/N ratio, Table 1), but no significant difference in either CO_2 fluxes or cumulative CO_2 emission at the NF compared to the RF and OP (Figure 5a), suggests that root respiration contributed to CO_2 emissions more than aerobic decomposition. However, due to the limited data available in this study regarding other environmental variables as well as the microbial community structure, more information is needed to better understand small-scale spatial variation in CO_2 emissions in the tropical swamp forests of peninsular Malaysia.

4.3. Nitrous Oxide (N_2O) Emissions

The wide range in N_2O fluxes at the OP compared to the NF (Figure 4b) indicates that nitrogen fertilizer inputs for palm oil production resulted in increased N_2O emissions [17,97,98]. The annualized N_2O emissions at the OP ($0.42 \text{ g N}_2\text{O m}^{-2} \text{ yr}^{-1}$) in this study were comparable to the emissions from drained mature oil palm plantations on Southeast Asian peatland ($0.12\text{--}2.52 \text{ g N}_2\text{O m}^{-2} \text{ yr}^{-1}$ [37,40,44]). Additionally, it was similar to the N_2O emissions from other types of agriculture on drained peatlands, such as Sago plantation: $0.33 \text{ g N}_2\text{O m}^{-2} \text{ yr}^{-1}$ [37], mixed-agriculture: $0.02 \text{ g N}_2\text{O m}^{-2} \text{ yr}^{-1}$ [17]. Cooper et al. [14] reported significantly higher N_2O emissions in young oil palm plantations ($29.1 \text{ g N}_2\text{O m}^{-2} \text{ yr}^{-1}$) that were likely explained by frequent applications of inorganic N fertilizer at an early stage of planting, as well as increased soil N levels due to growing leguminous cover crops, and high availability of decaying organic material from the recently cleared forest supplies labile C, which is essential for denitrifier heterotrophs.

However, another study by Oktarita et al. [99] found a minimal impact of fertilization-induced N_2O emission, with the majority of emission coming from decomposition in a field-based fertilization experiment on young oil palm. Similar findings were reported by Toma et al. [97] in croplands and a grassland on drained peat in Central Kalimantan. Furthermore, Kandel et al. [95] reported 0.15 to 3.77 of annual N_2O emission from rotating the type of croplands (e.g., rotations of oat–potato, oat–spring barley and potato–spring barley) at northern peatlands.

The N_2O emissions from natural wetland have frequently been found to be low but can have a high degree of variability [100,101]. Similarly, N_2O emissions in tropical peat swamp forest are generally low compared to emissions of CH_4 and CO_2 [5,37]. The annualized emission of N_2O at the undrained NF in this study ($0.08 \text{ g m}^{-2} \text{ yr}^{-1}$) was within the range of the observed emissions from Southeast Asian tropical swamp forests, e.g., Sarawak: $0.002\text{--}0.17 \text{ g N}_2\text{O m}^{-2} \text{ yr}^{-1}$ [37], Kalimantan: $0.06\text{--}0.52 \text{ g N}_2\text{O m}^{-2} \text{ yr}^{-1}$ [17,40]. Furthermore, Kandel et al. [95] recorded $0.03 \text{ g N}_2\text{O m}^{-2} \text{ yr}^{-1}$ from undrained natural bog of northern peatlands. Additionally, the annualized N_2O emissions at the rewetting-restored RF site in our study ($0.3 \text{ g N}_2\text{O m}^{-2} \text{ h}^{-1}$) were within the range of N_2O emissions in the rewetting tropical peatland system in Southeast Asia ($0.02\text{--}0.45 \text{ g N}_2\text{O m}^{-2} \text{ h}^{-1}$) reported in other studies [6,25,98].

The microbiological processes of nitrification and denitrification produce N_2O emissions from soils, with maximum emissions occurring in soils with intermediate aerobic-anaerobic conditions [102]. Thus, as denitrification and nitrification can occur concurrently in the same soil aggregate, the source of N_2O is frequently uncertain [102]. Linn and Doran [103] found linear relationships for N_2O production in incubated soils with WFPS values ranging from 30 to 70%, and maximum activity occurring at a 60% WFPS. Another study by Baral et al. [104] found that denitrification was accountable for 79–98% of N_2O emissions from aerobic conditions under 70% WFPS, and anaerobic conditions with 80% WFPS. The highest cumulative N_2O emission at the drained and fertilized OP and rewetting RF sites indicates ‘temporary denitrification’ or ‘aerobic-wet conditions’ (WFPS: 74.46 and 72.45, respectively, Table 1) may have occurred at the peat layer near to the surface. Hence, with low canopy cover at both sites (e.g., patches of open areas with less understory vegetation at the OP and less tall trees at the RF) allowing for a warmer soil surface (one degree higher than NF, Figure 2b) that could enhance soil organic matter mineralization, it provided inorganic N together with a high decomposition rate (low C/N ratio, Table 1) [105], which would have provided ideal conditions for microbial-mediated N_2O emissions [106,107].

The consistently small N_2O emissions at the NF (Figures 4b and 5b) can be attributed to a low decomposition rate (high C/N ratio, Table 1), limiting N availability for peat mineralization [108] and a low chemolithoautotrophic nitrification rate that was constrained by low oxygen availability [95] due to the continuously high WT level. Melling et al. [37] found that N_2O production differed with soil depth (e.g., highest at a 5-centimeter depth) in a tropical swamp forest ecosystem, which was largely caused by denitrification and restricted to flooded conditions all the time. Hence, as N_2O emissions in peatland were essentially derived from N mineralization in the oxidizing peat layer above the WT level [109], the rewetting of peat soil is likely to reduce N_2O emissions to levels comparable to natural, undrained peatlands. In accordance with this synthesis, Minkinen et al. [109] found that N_2O emissions from rewetted peatlands were similar to or lower than undrained, natural temperate peatlands in Finland, where there were slightly higher emissions in nutrient-rich (low C/N ratio) conditions than in nutrient-poor conditions. In contrast, the cumulative N_2O emissions from the rewetting RF system in our study were significantly higher than those of the undrained NF (Figure 5b). This may have been due to a high availability of N input from litter deposition and understory vegetation (low C/N ratio, Table 1), as well as a moderately high WT level (Figure 2d; averaged –18 cm depth). The same condition has been observed in another study of rewetting peatland in Kalimantan, Indonesia [25].

Soil N₂O emissions at a global scale are greatly influenced by changes in the environment, such as changes in rainfall distribution [110], where high rainfall may intensify the waterlogging and denitrification processes, increasing the rate of N₂O emission as a by-product of denitrification. The cumulative N₂O fluxes from the disturbed OP and RF peat were notably higher in August to early September (when the precipitation slightly increased, supplement data 1) possibly associated with either (i) gas diffusion or (ii) microbial activity (e.g., the inorganic N may occur in the thin water films of microsites near the subsoil (oxidizing layer), thus with the onset of a rainy day, the soil microbes can quickly use these N pools and instantly produce pulses of N₂O soon after a rainfall event [37].

4.4. Methane (CH₄) Emissions

The relative importance of CH₄ compared to CO₂ depends on its stronger warming potential, especially over shorter time periods. Due to the huge carbon pool and high WT level in tropical peatland, there is big potential for CH₄ emission to the atmosphere. Comparing the annual CH₄ emissions from the NF in our study (24.57 g CH₄ m⁻² yr⁻¹) to peatland in the peninsular Malaysia region, it was comparable to the CH₄ emissions (17.52 g CH₄ m⁻² yr⁻¹) recorded at Terengganu peat Selangor peat swamp forest [31], but three times higher than the CH₄ emissions reported by Cooper et al. [14] (7.6 g CH₄ m⁻² yr⁻¹) at North Selangor peat swamp forest. Additionally, the findings by Dhandapani et al. [31] resulted in an annual CH₄ uptake at North Selangor peat swamp forest ranging from -0.09 to -0.26 g CH₄ m⁻² yr⁻¹ in the wet and dry seasons, respectively. The CH₄ emissions from the NF natural forest in our study were much higher than the previously observed range for Southeast Asian swamp forests (3.8–5.6 g CH₄ m⁻² yr⁻¹ [79,111]), and also higher than fens in northern peatlands (7.5–15.6 g CH₄ m⁻² yr⁻¹ [112]). However, they were similar to the emission range recorded for bogs (15.7 g m⁻² yr⁻¹ [113], 20.5 g m⁻² yr⁻¹ [112], 39.4 g m⁻² yr⁻¹ [95]) in northern peatlands (Supplementary Table S3). The various reactions of CH₄ emissions in Southeast Asian peatlands could be related to regional peat features, e.g., coastal and inland peatlands in Sarawak and Kalimantan, respectively, which differ in age and carbon accumulation rates [114,115]. Additionally, in tropical peatlands, differences in aboveground vegetation can influence CH₄ emissions due to root exudate composition, because root exudates can provide large inputs of labile C, which can stimulate GHG emissions [116].

Peatlands are drained from the surface to reduce the WT, which has a large impact on the function of the peatlands. Reducing the WT through drainage changes the biological, chemical and physical characteristics of the soils, increasing soil aeration and soil temperatures, and thus reducing CH₄ emissions [117]. Annual soil CH₄ at the OP (1.09 g CH₄ m⁻² yr⁻¹) was higher than the CH₄ emissions from the drained agricultural peatlands reviewed by Hergoualc'h and Verchot [79] except for rice fields and Sago plantations. A temporal CH₄ emission study by Swails et al. [40] recorded 0.013 ± 0.7 g CH₄ m⁻² yr⁻¹ emissions in an oil palm plantation in Kalimantan, Indonesia. Additionally, Kandel et al. [95] found negligible CH₄ emissions from rotation croplands and recorded -0.15 to 0.15 g CH₄ m⁻² yr⁻¹ from drained permanent grassland on northern peatland bogs. Lowering the WT level at the OP would have limited the anaerobic zone of archaeal CH₄ production but increased the methane-oxidizing (methanotrophic) bacteria activity that uses CH₄ as an energy source [118], which would explain the CH₄ consumption in the unsaturated topsoil (>50% measurements, Figures 5c and S2c).

Peatland rewetting has been found to be a cost-effective GHG emission-cutting measure [119], although it re-establishes CH₄ emissions. Rewetting, as opposed to draining, increases the likelihood of creating anaerobic conditions in peat soils, which could activate methanogen bacteria. As a result of this, CH₄ can be produced and emitted. Some studies in northern peatlands have reported higher CH₄ emissions in rewetting peatlands compared with pristine peatlands [112,119–121]. The annualized CH₄ emissions at the RF site in this study were notably higher than the reported emissions in restored forest in Kalimantan, Indonesia [25] and the emission values for rewetted organic soil tropical peatland set by the

IPCC [1] (default value of $4.1 \text{ g CH}_4 \text{ m}^{-2} \text{ yr}^{-1}$), but lower than the emissions for natural wetland tropical swamp, which are $29.7 \text{ g CH}_4 \text{ m}^{-2} \text{ yr}^{-1}$ (Supplementary Table S2).

The emission of CH_4 in our study was consistently greater at the RF and NF forested sites (Figure 4c) than at the drained OP site, which was likely due to the relatively high WT (Figure 2d), which never dropped below -45 cm , and these findings complement previous studies showing that CH_4 emissions increased significantly with a rise in the WT level above -20 cm [22,122]. Waterlogging of the surface layers results in anoxic conditions that are favorable for anaerobic respiration and CH_4 production in peat swamp forest. At the NF site, CH_4 emissions increased in magnitude and became more variable compared to the other two sites (Figures 4c and 5c) as the WT level approached the soil surface (Figure 2c), indicating the presence of a threshold response to WT level, similar to that observed by Jauhainen et al. [123]. The cumulative CH_4 emissions at the RF were lower than at the NF site (Figure 5c), potentially due to the lower WT level (mean: -18.6 cm and -6.0 cm at the RF and the NF, respectively), because a deeper WT would have increased the oxygenation of surface peat layers, which would have encouraged methanotrophy over methanogenesis, as previously observed in studies of Southeast Asian peatlands [4,96,123]. Furthermore, the high CH_4 fluxes found between the end of March and early June 2018 seem to be associated with an increasing WT (Figures 2d and 5c), which is the phenomena known as the 'rewetting effect' [124]. The following two mechanisms have been commonly acknowledged as causing CH_4 pulses after reflooding: (i) the enhancement of microbial metabolism due to the available substrates during the drained period and (ii) gas displacement in soil, which is the mechanism more likely to have taken place in our forested sites.

In contrast to our finding, Dhandapani et al. [31] reported that CH_4 oxidation at a north Selangor swamp forest was influenced by soil moisture limitations (52–55%), which was slightly lower than our forested sites (56–61%) and those of a study by Cooper et al. [14] (82.3%). Additionally, their recorded WT level was below the surface through the year (50–60 cm during the dry season). The contradiction between our finding and those of Dhandapani et al. [31] can be explained by the different monitoring locations, which correspond to different environmental conditions and peat characteristics. Dhandapani et al. [31] samples collection was located at the northern edge of the NSPSF, within the Sungai Karang Forest Reserve, whereas our data were collected at Raja Musa Forest Reserve, on the southern edge of the NSPSF (Figure 1). The findings of Dhandapani et al. [31] and our study are important to fill the gap of understanding the C budget in the NSPSF. Other studies at Sarawak and Kalimantan have reported positive correlations between CH_4 emissions and soil moisture [45,122]. The WT level regulates soil moisture, which is critical in regulating the CH_4 emissions from tropical peat swamp forest. High soil moisture increases the thickness of the anaerobic zone that promotes methanogenesis and boosts CH_4 production in the soil layer [125]. The similar moisture values recorded at the RF and the NF (Figure 2c) indicate that soil moisture may have contributed to the CH_4 emission at the RF recovery forest.

Studies in temperate and boreal peatlands have revealed a positive relationship between CH_4 emissions and soil temperature [126,127], as has a microcosm study by Wang et al. [128] for the mountain peatland. However, with a narrow seasonal range ($\sim 2 \text{ }^\circ\text{C}$) in soil temperatures in tropical peatland, comparing natural (NF) and disturbed peat (RF and OP) does not contribute to the CH_4 emissions. Furthermore, other studies have mentioned that the changes in tropical peatland CH_4 emissions will be driven more by shifting rainfall patterns and land management on peat hydrology than by rising temperatures [129].

4.5. Global Warming Potential (GWP)

The concept of global warming potential has been introduced by the IPCC to convert non- CO_2 emissions (e.g., CH_4 and N_2O) to CO_2 -equivalents with the aim of comparing the overall radiative forcing of all greenhouse gases. In the current study, the higher 20-year GWPs for the total GHG emissions from the forested system NF and RF sites were

higher than the GWP for the oil palm plantation site, mainly due to greater CH₄ emissions that contributed more than 50% of the total GWP. Previous studies have also shown that recovering peatlands by rewetting in both tropic and temperate regions can increase CH₄ emissions [23,109,130], with the amount of CH₄ emissions related to the peat quality and high WT level. The CO₂ emissions in the OP contributed 78% of the total 20-year GWP for that site; hence, other studies of matured oil palm plantation (more than 10-years old) on tropical peatland have shown that more than 50% of their 20-year GWP owned to CO₂ emissions (Table 3). Persistently high CO₂ emissions from drained oil palm plantations is promoted by lowering the WT level; thus, consistently increasing the decomposition of organic matter. The CO₂ emissions from the drained OP site are significantly lower than the average 6000 g m⁻² yr⁻¹ prescribed by the Roundtable on Sustainable Palm Oil (RSPO) [131] for sites with WT levels within 40 to 60 cm of the soil surface [14]. This might be explained by the plantation intensity system, where the commercial plantation may contribute more to GHG emissions than a smallholder's plantation. Despite the high contribution of CO₂ emissions in the OP, the total 20-year GWP was still much lower than at the NF or RF sites in this study, due to their higher CH₄ emissions. The influence of N₂O emissions on the 20-year GWP was low from the drained OP (12%), but minor in forested sites (4 and 1% in the RF and the NF, respectively). A previous study has shown that high N₂O emissions were contributed during the early conversion of oil palm (71%) [14]. Drained forest and early oil palm conversion were the highest contributors to the 20-year GWP (110,486 ± 22,88–25,853 ± 75.43g CO₂ eq.m⁻² yr⁻¹) followed by matured plantation and recovery and primary forest systems (Table 3).

5. Conclusions

In conclusion, we have demonstrated temporal measurements of GHGs emission for three different land-use changes in tropical peatland in Peninsular Malaysia. Our results indicate that the significant differences in the environmental conditions of the drained, rewetting-restored and natural peatland systems affected CH₄ and N₂O the most, with little effect on CO₂ emissions. Tropical peat characteristics are altered by oil palm agriculture compared to forested systems, but the overall measured properties did not vary significantly between land uses. However, there were notable differences in the C/N ratio and decomposition rates between the peats that had been drained (currently drained OP and historically drained RF) and the undrained natural peat (NF), which were associated with differences in the cumulative CO₂ and N₂O emissions. Methane emissions were significantly increased in the recent rewetting-restored RF peatland compared with the still-drained OP peatland due to the raised WT level, but this had no significant effect on reducing CO₂ emissions due to understory vegetation, which had a higher proportion of root respiration as well as decomposition of the available labile C. When emissions of the three main GHGs were accounted for over a 20-year, the highest total GWP in the forested system was due to CH₄ emissions. Research on how peatland ecosystem rehabilitation strategies affect the GHG emissions, and its mechanisms are critically required. Based on our findings, we can assert that the rewetting-restored efforts of tropical peatland may or may not reduce GHGs depending on the timeline after they started recovering. Although the initial stage of peatland recovery (e.g., less than seven years) may not be possible to reduce GHGs, further research in accounting for the GHGs from stages of rewetting-restored efforts (e.g., timeline base) need to be explored. In addition, restored peatlands should broaden the context (e.g., different types of restored sites and the history of the previous land-used before implementing a rewetting-restored effort). This knowledge can help us to better understand the beneficiaries of restored efforts in reducing GHGs and recovering peat properties in the longer term. Additionally, this information may aid state forestry to make a better decision on forest management practices.

Supplementary Materials: Supplementary data to this article can be found online at <https://www.mdpi.com/article/10.3390/w13233372/s1>, Figure S1: Rainfall data from 3 weather stations, Batang Berjuntai, Sungai Karang and Sungai Tenggi Kiri. Error bars represent standard error of the mean,

Table S1: GHGs emission of drained-agricultural in tropical peatlands, tropical mineral and northern peatland by closed chamber and estimation from subsidence measurement, Table S2: GHGs' emissions of managed, recovery-rewetting, secondary-drained forest systems in tropical and northern peatlands by closed chamber and estimation from subsidence measurement, Table S3: GHGs emission of natural forest system in tropical and northern peatlands by closed chamber and estimation from subsidence measurement.

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