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Significant sedge-mediated methane emissions from degraded tropical peatlands

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Abstract

Sedge-mediated gas transport to the atmosphere has been recognized as a significant CH₄ pathway in northern peatlands; however, in the Tropics, this pathway remains unquantified. In Southeast Asia, degraded tropical peatlands covered with sedges and ferns have increased to approximately 10% of the total peatland area due to an increased drainage and fires. In view of this, we investigated the role of sedge, Scleria sumatrensis, in CH₄ emissions from a fire-degraded tropical peatland in Brunei. At our site, we found that this sedge-mediated transport contributed >70% of the total CH₄ emission, making it a significant CH₄ emission pathway. We also observed significant seasonal and spatial variation with values ranging from 0.78 ± 0.14 to 4.86 ± 0.66 mgCH₄ m⁻² h⁻¹. This variation was mainly attributed to water table level along with changes in sedge cover and pore-water properties (pH, salinity, cations, and anions). More importantly, these numbers are three times higher when compared to intact peat-swamp forests and 17 times higher when compared to similar degraded tropical peatland covered with shrubs.

1. Introduction

Tropical peatlands in Southeast (SE) Asia store around 68.5 Gt C, which equates to 11–14% of the total global peat organic carbon pool [1]. Despite being one of the world’s richest C reserves, tropical peatlands have been deforested and drained for land development over the past few decades [2–4]. These disturbances lead to drier peat surface [5], making these areas vulnerable to fire, and eventually resulting in loss of vegetation and significant C emissions [6]. For reference, the fire event in 1997–1998 burnt 0.73 Mha of peatlands in SE Asia thereby releasing 0.81–2.57 Gt C [7], whereas the September–October 2015 episode burnt 2.6 Mha releasing 0.23 Gt C from southern Sumatra and Kalimantan alone [8, 9, 10]. The most recent September–October 2019 fire event burnt approximately 0.86 Mha of peatlands in Indonesia, resulting in 0.36 Gt C emissions [8, 10]. More importantly, the extent of such degraded peatland areas has increased to almost one-tenth of the total peatland area in SE Asia (~1.42 Mha) [4].

Fire not only results in C emissions but also affects peatland vegetation composition, peat, and pore-water chemistry. Specifically, fire destroys the original vegetation leading to degraded open-peatland areas occupied by pioneering species such as ferns and sedges [5, 11]. These plants, together with drier peat surface, have a higher chance of fire ignition during periods of extensive drought [12, 13]. Furthermore, the possibility of fire reoccurrences afterward becomes higher in these ferns and sedge dominated areas [5, 14]. Besides, fire also affects peat characteristics such as bulk density, quantity, and quality of organic matter, nutrient availability, and pore-water chemistry. All these can influence the recovery of
degraded peatlands to secondary peat-swamp forests (PSFs) [11].

Peat hydrology is also significantly affected by fire due to the loss of surface C and drainage induced peat subsidence, where latter can result in average loss of 100 t CO₂ eq ha⁻¹ yr⁻¹ [15]. Both these activities result in peat surface lowering, which can lead to an increased risk of flooding during the wet season [11, 16]. In degraded tropical peatlands, the total flooded area can expand up to five times with increasing water table (WT) level, as shown in a 2006 study in Jambi province, Indonesia [17]. The same study predicted that during wet period, with prolonged groundwater levels >1 m above the surface, flooding would inhibit regrowth of original plant species. Instead, more flood-tolerant plant species such as sedges might thrive [11, 17], and these are recognized as a significant source of CH₄ in northern peatlands [18, 19]. However, in case of tropical peatlands, this sedge-mediated CH₄ emission pathway remains unquantified.

The stems of sedges, due to the presence of aerenchyma tissues, can serve as a chimney for CH₄ transport from the anoxic zone to the atmosphere, thus bypassing its oxidation in the oxic-zone [20]. In northern peatlands, CH₄ emission from sedges may contribute ∼28–90% of the total CH₄ flux [18]. In the tropics, specifically in SE Asia, given that the extent of open-degraded peatlands occupied by ferns and sedges accounts for ∼10% of the total peatland area (∼1.42 Mha) [4], it is crucial to investigate further this potentially important source of CH₄, which has a global warming potential 34 times more than CO₂ [21]. Many studies have focused more on CO₂ emissions from degraded peatlands area with low WT [22–26], however, with sustained CO₂ emissions and resulting subsidence, these areas becomes susceptible to recurrent floods, creating conditions conducive for methanogenesis, and hence higher CH₄ emissions [16, 24].

To address this critical knowledge gap, we examined the effects of vegetation composition, environmental variables, and pore-water characteristics on CH₄ emission from a fire-degraded tropical peatland. We hypothesize that in fire-degraded tropical peatland: (1) sedges are a significant source of CH₄, transporting it directly into the atmosphere as in northern latitudes; (2) fire induced changes in local hydrology, peat and pore-water properties can affect the sedge cover and hence CH₄ emissions.

Our findings will reduce the uncertainties in quantifying CH₄ emissions from degraded tropical peatlands and hence refine the regional CH₄ budget.

2. Material & methods

2.1. Study area location and site description
The fire-degraded tropical peatland study area lies 20 km south of the coast in Belait District, Brunei Darussalam (4°28’40″ N, 114°18’19″ E; figure 1). The region experiences an equatorial-humid climate with two seasons, separated by two transitional periods (two maxima and two minima) [27]. The first maximum and second minor maximum lasts from late October to January and from May to June, respectively, corresponding to the wet period (ibid). The lowest minimum and minor minimum occur from February to March and June to August, respectively, corresponding to the dry period (ibid). Average rainfall during the wet and dry period (for 1966–2006) is 1939.4 ± 51.6 mm and 924.8 ± 46.3 mm, respectively (ibid). The average monthly temperature is 27.2 °C ± 0.4 °C, which varies annually between 23.4 °C and 30.8 °C [27, 28].

We identified our study area from LANDSAT imagery using two criteria. Firstly, the area had burnt multiple times (n = 7; 1998, 2005, 2009–2010, 2014–2016) over the last few decades and had never recovered back to secondary PSF state [24], and therefore, the study area was covered with pioneering and flood-tolerant plant species such as ferns and sedges. Secondly, due to the disruption of original hydrology by the construction of an access road and drainage canal, the area closer to the drainage canal remained relatively water-saturated all-year round and hence covered with sedges.

2.2. Experimental design
Our fire-degraded tropical peatland study area stretches 2 km laterally along the road with drainage canal. To cover this spatial variability, we established three sites (B1–3) at an interval of 600–800 m along the road, adjacent to the drainage canal. Each site had two plots of 5 m × 5 m, i.e. 2 m and 10 m away from the drainage canal (figure 1). The choice of these specific plots was based on the presence of sedges all year-round, hence allowing continuous CH₄ measurements during the dry and wet period.

The study area also stretches 300–500 m east-west, towards the forest-edge. Hence, to capture the spatial variability away from the drainage canal, we selected a longer transect (T4; described in Lupascu et al 2020), where plots (of 5 m × 5 m) were at an interval of 100 m from the drainage canal towards the forest (B2–T4; figure 1).

In each plot, we recorded environmental variables, pore-water physicochemical properties, CH₄ emissions, and peat characteristics (supplementary information (available online at https://stacks.iop.org/ERL/16/014002/mmedia)). These observations were recorded over four fieldwork campaigns, i.e. in August 2018, October 2018, January 2019, and July 2019.

2.2.1. Environmental variables
Air temperature and barometric pressure (Barodiver, VanEssen Instruments, Netherlands) were measured continuously close to the drainage canal on
Figure 1. (a) Study area location in Brunei Darussalam. (b) B1−3 are three sites, each having plots (of 5 m × 5 m) at 2 m away (B1a, B2a, B3a) and 10 m away (B1b, B2b, B3b) from drainage canal and T1−5 are the transects ranging up to 300 m. (c) Schematic of quadrat placement in plots on transects (T1−5) used for vegetation composition assessment. Additionally, plots at the T4 transect (100 m, 200 m, and 300 m) were used for CH$_4$ flux measurements.

Figure 1.

2.2.1. Environmental variables

Soil temperature and moisture were measured in three plots (B1a, B2a, B3a) at 2 m away from the drainage canal and 10 m away (B1b, B2b, B3b) for every fieldwork campaign (in August 2018, October 2018, January 2019, and July 2019), using a YSI556 Multiprobe (YSI Life Sciences, USA). Water table (WT) levels were monitored with pressure transducers (TD-Diver, VanEssen Instruments, Netherlands) installed in piezometers made of perforated PVC pipes (1.5−2.4 m deep, depending on the peat depth). These environmental variables were monitored from August 2018 to July 2019. Precipitation data was taken from a nearby (about 30 km) weather monitoring station, Sungai Liang, Belait District, Brunei Darussalam [28].

2.2.2. Physicochemical analysis of pore-water

pH, Temperature, Dissolved Oxygen (DO), Electrical Conductivity (EC), Total Dissolved Solids (TDS), and Salinity were measured on-site in the pore-water (10 cm depth) of piezometers installed in each plot (n = 6) for every fieldwork campaign (in August 2018, October 2018, January 2019, and July 2019), using a YSI556 Multiprobe (YSI Life Sciences, USA). For Dissolved Organic Carbon (DOC) and ions analysis, pore-water samples were collected using a push-point sampler in cleaned 50 ml polypropylene bottles (Falcon vials, Fisher Scientific, USA). Samples were filtered using 0.45 micron PTFE filters (Whatman 25 mm GD/X Syringe Filters), stored in the cool box during transportation, and then refrigerated at 4°C until further analysis. DOC was analyzed using a Total Organic Carbon analyzer (Vario TOC-Cube, Elementar), whereas concentration of anions (Cl$^-$, NO$_2^-$, NO$_3^-$, SO$_4^{2-}$, PO$_4^{3-}$) and cations (Na$^+$, K$^+$, NH$_4^+$, Mg$^{2+}$, Ca$^{2+}$) were analyzed with Ion Chromatograph (ICS-5000, Dionex) in the Geolab, Department of Geography, NUS.

2.3. Vegetation composition assessment

The vegetation at our site was mainly composed of vascular plants. A 1 m × 1 m quadrat [n = 47 in total including both dry (n = 22) and wet (n = 25) period] was used to estimate vegetation per cent cover (number of small 10 cm × 10 cm grids occupied by plant species within 1 m × 1 m quadrat), height, and the total number of plant species in the quadrat [29]. The
assessment was conducted on five transects (T1—5; figure 1), where measurements were recorded at 2 m, 5 m, 10 m, 50 m, 100 m, 200 m, and 300 m from the drainage canal. We used this non-uniform sampling design since we observed a noticeable shift in plant communities (more sedges and fewer ferns) within this short interval compared to plots towards forests (50 m, 100 m, 200 m, and 300 m; figure 1(c)) [30].

To capture the seasonal variation, vegetation composition assessments were conducted during the wet (in January 2019) and dry (in July 2019) period. However, due to a fire event in February 2019, which burnt two transects (T3—4), nearly four months before the July 2019 assessment, we analyzed these data separately from the ‘previously’ burnt transects (T1, T2, T5—burnt in March 2016).

2.4. Measurement of methane flux
We installed 3—4 permanent PVC collars (of 23 cm diameter) per plot for CH4 flux measurements on the peat surface (without vegetation). These collars were installed in June 2018, two months before the first measurement (August 2018), and inserted 15—20 cm deep to exclude any root-regrowth. For the total CH4 flux measurement, i.e. with vegetation (sedges + peat), 9—12 PVC collars (of 10 cm diameter) per plot were inserted few centimeters deep in peat to avoid root damage but to ensure proper sealing for flux measurement.

CH4 flux was measured using a portable Greenhouse Gas Analyzer (GasScouter, Picarro Inc. Santa Clara, CA, USA) connected to a transparent chamber. The chamber was made of a clear acrylic pipe of 11 cm internal diameter and variable height up to 200 cm resulting in a maximum total volume of 19.1 l. Gas collected inside the chamber was mixed using a battery-operated fan, before passing through 1.5 m tubing (3 mm internal diameter) to the gas analyzer and circulated back to the chamber (figure S2). To estimate the sedged-mediated CH4 emissions, measurements without vegetation (peat-only) was recorded adjacent to measurements with vegetation (i.e. sedge + peat) and subtracted from it to estimate the net CH4 contribution from sedges (n = 16—63 per field campaign). All measurements were recorded between 09:00 and 17:00 h. The chamber was left closed for up to 5 min, and CH4 flux (F) was calculated as per the equation below:

$$F = \frac{s \times V \times t}{R \times T_{\text{air}} \times A}$$

where $F$ is in mgCH4 m$^{-2}$ h$^{-1}$; $s$ is the linear regression slope of CH4 concentration change over time in ppm per second (accepted with $r^2 > 0.85$ and $p < 0.05$) [31, 32]; $V$ is the volume of the chamber, tubing, and the internal cavity of the gas analyzer in liters; $t$ is the conversion factor from seconds to hours, $\mu$g to mg, and 16 $\mu$g $\mu$mol$^{-1}$ of CH4; $R$ is the universal gas constant (8.205 $\times$ 10$^{-5}$ 1 atm $\mu$mol$^{-1}$K$^{-1}$); $T_{\text{air}}$ is the air temperature inside the chamber (K); $A$ is the surface area of the chamber (m$^2$).

With each flux measurement, soil temperature (T) and soil water content (SWC) were measured beside the collar using a digital thermometer (15—077, ThermoFisher Scientific, USA) and soil moisture meter (Dynamax TH300, USA) at a depth of 5 cm.

2.5. Statistical analysis
All the statistical tests were performed using Origin-Pro (2019), and significance was accepted at $p < 0.05$. The data was checked for normality using the Shapiro—Wilk test. Accordingly, non-parametric tests, namely Kruskal–Wallis ANOVA and Mann–Whitney U tests, were used for data comparison. Spearman correlation coefficients were calculated to evaluate the relationship between CH4 emission and number and height of sedges, environmental variables, and pore-water properties. All the values are reported as mean ± s.e. unless stated separately as mean ± s.d.

3. Results

3.1. Environmental variables
Environmental variables were recorded from August 2018 to July 2019 with a total of 337 observation values ($n = 173$ for the dry period and $n = 164$ for the wet period). Air temperature over the sampling period showed an overall annual mean of 26.9 $^\circ$C ± 1.6 $^\circ$C with an average monthly value of 27.1 $^\circ$C ± 1.5 $^\circ$C in the dry period and 26.7 $^\circ$C ± 1.7 $^\circ$C in the wet period (figure 2(a)). Similarly, peat soil temperature showed a mean of 27.4 $^\circ$C ± 1.0 $^\circ$C for the study period, but with no significant difference between dry (27.4 $^\circ$C ± 0.9 $^\circ$C) and wet (27.3 $^\circ$C ± 1.1 $^\circ$C) period (figure 2(b)).

Total annual precipitation over the sampling period was 1447.8 ± 12.5 mm, with 505.9 ± 10.2 mm and 941.9 ± 14.3 mm during the dry and wet period, respectively. Precipitation increased from August 2018 (75.1 mm) until January 2019 (190 mm); however, it remained infrequent from February 2019 to July 2019 (figure 2(c)).

The WT was affected by precipitation patterns, with higher values during the wet period (−7.1 ± 10.2 cm), compared to the dry period (−26.0 ± 10.0 cm; figure 2(c)).

Soil water content varied from 78.8 ± 13.4% to 91.6 ± 14.4% during the dry and wet period, respectively.

3.2. Physicochemical properties of pore-water
We observed seasonal differences in pore-water properties where mean values for pH, Temperature, EC, TDS, Salinity, and DOC, except DO, were overall higher during the dry period (August 2018 and July 2019) compared to the wet period (October 2018 and
January 2019) (table 1). However, only pH and Salinity differed significantly \( (p < 0.05) \), with a decrease in pH from \( 4.1 \pm 0.2 \) to \( 3.1 \pm 0.3 \), and in Salinity from \( 0.032 \pm 0.003 \% \) to \( 0.024 \pm 0.001 \% \) during the dry and wet period, respectively. Comparing the data from before the fire event in February 2019 with our July 2019 sampling, we observed an increase for all parameters except for DO.

Pore-water ions concentration remained stable for the first three sampling campaigns between August 2018 and January 2019 (table 2) with Cl\(^-\) ranging from 3.53 to 4.31, SO\(_4^{2-}\) from 0.21 to 0.34, PO\(_4^{3-}\) from 0.26 to 0.39, Na\(^+\) from 2.30 to 3.71, K\(^+\) from 0.87 to 0.96 mg l\(^{-1}\). Ion concentrations were significantly \( (p < 0.05) \) higher in our July 2019 sampling compared to previous months. Specifically, when compared to Jan 2019 values, the concentrations of Cl\(^-\) and SO\(_4^{2-}\) were five times, PO\(_4^{3-}\) concentration was eight times, and Na\(^+\) and K\(^+\) concentrations were three and seven times higher, respectively (table 2).

### 3.3. Vegetation composition assessment during the wet and dry period

During our wet (January 2019) and dry (July 2019) period assessment, we recorded only one species of sedge (Scleria sumatrensis) and three species of ferns (Blechnum indicum, Nephrolepis hirsutula, Nephrolepis biserrata).

We observed that the vegetation cover change was spatially significant \( (p < 0.001) \) compared to seasonal variability, where the mean sedge cover increased from 5% to 9% in plots far from the drainage canal to 23% to 31% in plots near the drainage canal (figures 3(a) and (b)).

We also analyzed the vegetation data separately for previously (in March 2016) and recently (in February 2019) burnt transects to observe the effects of recent fire episodes on the number of sedges. However, similar to the above-mentioned, we found that the change in the number of sedges was spatially significant \( (p < 0.001) \) compared to seasonal variability or fire episode. Therefore, using the average of both the dry and wet period, we found that the number of sedges along the transect declined from 28 sedges per m\(^2\) in plots near canal (plots at 2 m, 5 m, and 10 m) to 14 sedges per m\(^2\) in plots towards forest (plots at 50 m, 100 m, 200 m, and 300 m) on previously burnt transect (figure 4(a)). Similarly, it declined from 43 sedges per m\(^2\) in plots near canal to 5 sedges per m\(^2\) in plots towards forest on recently burnt transect (figures 4(b)).

The average height of sedges decreased from \( 1.6 \pm 0.3 \) (\( n = 125 \)) m to \( 1.3 \pm 0.2 \) (\( n = 110 \)) during the wet and dry periods, respectively.

### 3.4. CH\(_4\) emissions

We found that the sedges in degraded tropical peatlands are a significant source of CH\(_4\), transporting approximately >70% of the total CH\(_4\) emissions directly into the atmosphere (tables S1 and S2). This
sedge-mediated transport varied seasonally as well as spatially during the sampling period.

Over the sampling period, sedge-mediated \( \text{CH}_4 \) emissions contributed to an annual mean of 3.77 ± 0.47 mg CH\(_4\) m\(^{-2}\) h\(^{-1}\), compared with a total flux of 4.61 ± 0.44 mg CH\(_4\) m\(^{-2}\) h\(^{-1}\) \((n = 180)\) (Inset figure 5). CH\(_4\) emissions significantly \((n = 180; p < 0.001;\) Kruskal–Wallis ANOVA, \( H = 67.84, DF = 3\)) varied over the sampling period following rainfall patterns, and hence WT levels. Lower values were recorded during the dry period with 1.60 ± 0.47 and 0.67 ± 0.08 mg CH\(_4\) m\(^{-2}\) h\(^{-1}\) for August 2018 and July 2019, respectively. The peak in CH\(_4\) emission was observed in the wet period with 3.52 ± 0.83 and 5.96 ± 0.96 mg CH\(_4\) m\(^{-2}\) h\(^{-1}\) in October 2018 and January 2019, respectively (table S1). These net emissions were approximately five times higher compared to the dry period.

We also observed spatial variation in sedge-mediated CH\(_4\) emission where plots near the drainage canal (at 2 m, 5 m, and 10 m) emitted higher CH\(_4\) compared to plots farther away from drainage canal, i.e. towards the forest edge (plots at 50 m, 100 m, 200 m, and 300 m) (figure 6). This spatial variation was significant \((n = 180; p < 0.001;\) Kruskal–Wallis ANOVA, \( H = 42.48, DF = 5\)) and

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**Table 1.** Physicochemical properties of pore-water (mean ± s.e.).

| Samples                     | pH      | Temp. (°C) | DO (mg l\(^{-1}\)) | EC (μS cm\(^{-1}\)) | TDS (mg l\(^{-1}\)) | Salinity (%) | DOC (mg l\(^{-1}\)) |
|-----------------------------|---------|------------|---------------------|----------------------|----------------------|--------------|---------------------|
| August 2018 \((n = 6)\)    | 3.7 ± 0.1 | 25.7 ± 0.2 | 0.8 ± 0.2           | 62.2 ± 4.1           | 40.3 ± 2.6           | 0.027 ± 0.002 | 62.7 ± 2.3         |
| October 2018 \((n = 6)\)   | 3.2 ± 0.5 | 26.6 ± 0.3 | 3.2 ± 0.5           | 56.7 ± 6.2           | 37.8 ± 4.5           | 0.023 ± 0.002 | 38.6 ± 5.9         |
| January 2019 \((n = 6)\)   | 3.0 ± 0.2 | 25.7 ± 0.2 | 0.8 ± 0.4           | 58.7 ± 4.7           | 40.0 ± 8.9           | 0.025 ± 0.002 | 62.6 ± 2.3         |
| July 2019 \((n = 4)\)      | 4.7 ± 0.3 | 27.2 ± 0.2 | 1.3 ± 0.5           | 77.5 ± 7.5           | 57.5 ± 9.5           | 0.040 ± 0.004 | 65.0 ± 5.7         |

\(^{a}\)Pore-water in two of the wells (at site B3, i.e. in plots B3a & B3b) was very high in dissolved solids; therefore samples for these could not be collected. n.a is not available. bdl is below the detection limit.

**Table 2.** Comparison of pore-water major macro-nutrients (mean ± s.d.; values in mg l\(^{-1}\)).

| Ions     | August 2018 \((n = 6)\) | October 2018 \((n = 6)\) | January 2019 \((n = 6)\) | July 2019 \((n = 4)\) |
|----------|-------------------------|--------------------------|--------------------------|-----------------------|
| Cl\(^{-}\) | 3.53 ± 1.62             | 2.34 ± 1.78              | 4.31 ± 1.22              | 16.39 ± 3.13          |
| NO\(_2\)\(^{-}\) | 0.08 ± n.a              | bdl                      | 0.11 ± n.a              | bdl                   |
| NO\(_3\)\(^{-}\) | 0.13 ± 0.01            | 0.16 ± n.a              | 0.15 ± 0.02             | bdl                   |
| SO\(_4\)\(^{2-}\) | 0.21 ± 0.05           | 0.34 ± 0.11             | 0.34 ± 0.24             | 1.20 ± 0.26           |
| PO\(_4\)\(^{3-}\) | 0.26 ± 0.07           | bdl                      | 0.39 ± n.a             | 2.32 ± 1.58           |
| Na\(^{+}\) | 3.55 ± 2.09            | 2.30 ± 1.08             | 3.71 ± 1.58             | 9.72 ± 6.44           |
| K\(^{+}\) | 0.96 ± 0.21            | 0.87 ± 0.05             | 0.87 ± 0.03             | 6.00 ± 1.61           |
| NH\(_4\)\(^{+}\) | bdl                    | 0.07 ± 0.06             | bdl                      | 1.25 ± 1.69           |
| Mg\(^{2+}\) | 0.78 ± 0.19            | 0.60 ± 0.14             | 0.68 ± 0.08             | 0.72 ± 0.2            |
| Ca\(^{2+}\) | 1.68 ± 0.68            | 1.41 ± 0.58             | 1.54 ± 0.53             | 1.82 ± 0.38           |

\(^{a}\)Significant difference between seasons (at \( p < 0.05 \)).

\(^{b}\)Pore-water in two of the wells (at site B3, i.e. in plots B3a & B3b) was very high in dissolved solids; therefore samples for these could not be collected. n.a is not available. bdl is below the detection limit.

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**Figure 3.** Vegetation cover change during the dry (July 2019) and wet (January 2019) period in the plots: (a) far from the drainage canal (plots at 50 m, 100 m, 200 m, and 300 m), and (b) near the drainage canal (plots at 2 m, 5 m, and 10 m).
Figure 4. Number of sedges per m$^2$ along the transects on (a) previously burnt (T1, T2, and T5) and (b) recently burnt transects (T3–4). The higher number of sedges was present in plots near the drainage canal, i.e., plots at 2 m, 5 m, and 10 m compared to plots away from the drainage canal (towards the forest boundary) at 50 m, 100 m, 200 m, and 300 m away.

Figure 5. Seasonal variation in sedge-mediated CH$_4$ emissions and water table level during the dry (August 2018 and July 2019) and wet period (October 2018 and January 2019) (table S1). The inset graph shows an overall mean for CH$_4$ flux over the sampling period.

was mainly attributed to the higher number of sedges and higher WT in plots closer to the drainage canal. Sedge-mediated CH$_4$ emissions from plots near the drainage canal (2 m, 5 m, 10 m) emitted a mean of 3.48–5.41 mgCH$_4$ m$^{-2}$ h$^{-1}$, whereas plot farther away (at 100 m, 200 m, 300 m) emitted a mean of 0.63–1.75 mgCH$_4$ m$^{-2}$ h$^{-1}$ (table S2).

Overall, the CH$_4$ fluxes measured during both dry and wet period showed a significant positive correlation with the number ($p < 0.001$) and height of sedges ($p < 0.05$), soil water content ($p < 0.001$) and WT level ($p < 0.001$), and a significant negative correlation with distance from drainage canal ($p < 0.001$) (table S3). Among these, the WT level showed the strongest relationship with CH$_4$ ($r^2 = 0.78; p < 0.001$). In addition, pore-water parameters such as pH ($p < 0.001$) and Salinity ($p < 0.05$) were significantly correlated with CH$_4$ emissions (table S4).

4. Discussion

4.1. Role of sedges in CH$_4$ emissions

Our analysis show for the first time that sedge-mediated CH$_4$ transport to the atmosphere from degraded tropical peatland contributes approximately 70–80% of the total CH$_4$ emission at an ecosystem level. This result is similar to plant-mediated CH$_4$ emissions in northern peatlands, which transports ~28% to >90% of the total CH$_4$ emission to the atmosphere at an ecosystem level [18, 20, 33].
In the Tropics, few studies have investigated the tree-mediated CH$_4$ transport in the intact tropical peatlands of Panama [34, 35], Brunei [36], and Amazon flood plains [37], where such plant-mediated transport contribute in the range of $\sim$30–92% of the total CH$_4$ emission to the atmosphere at an ecosystem level.

Considering that degraded tropical peatlands cover approximately one-tenth of the total peatland area in SE Asia (as in 2015 [4]), it is evident that the contribution of sedge-mediated CH$_4$ emission is an important factor in refining the regional CH$_4$ budget. Although the percentage of sedge-mediated CH$_4$ emission remained consistent during the dry (70%) and wet (80%) period, the CH$_4$ emission values were significantly lower during the dry period ($0.78 \pm 0.14$ mgCH$_4$ m$^{-2}$ h$^{-1}$) when compared to the wet period ($4.86 \pm 0.66$ mgCH$_4$ m$^{-2}$ h$^{-1}$), which is due to significant positive correlation with the WT level (figure 5). In fact, studies have reported that a WT lowering of 20 cm below surface disrupts the optimum redox conditions for methanogenesis as well as exposes the anoxic peat to aerobic decomposition [38–40]. It is worth noting that the CH$_4$ emitted via sedges may primarily be originated from recently fixed carbon (litter, root exudates) and to lesser extent from bulk peat as confirmed from recent studies [24, 41]. Although this recently fixed C is considered as a short-term cycling C [42], nevertheless the significant CH$_4$ emission from sedge-mediated transport additionally contribute to the existing sources of CH$_4$ from degraded tropical peatlands, which are expected to increase in future.

We also observed a significant spatial variation in sedge-mediated CH$_4$ emission where values were four times higher in the area near the drainage canal ($4.33 \pm 0.56$ mgCH$_4$ m$^{-2}$ h$^{-1}$) compared to the area far from the drainage canal, i.e. towards the forest edge ($1.12 \pm 0.24$ mgCH$_4$ m$^{-2}$ h$^{-1}$). Although burning did affect the number of sedges along the transect, the differences were not statistically significant. Moreover, this may have been compensated by fast regrowth of sedges in the presence of increased nutrients released after the fire (table 2) as well as constant high soil water content (72–92%) in these plots [43, 44]. Nevertheless, with future frequent flooding and changes in vegetation composition of such fire-degraded tropical peatland areas, these emission values are likely to shift towards the higher end.

4.2. Effects of environmental variables and pore-water properties on CH$_4$ emissions

Our study showed that sedge-mediated CH$_4$ emission correlated with the WT, temperature, and pore-water properties (tables S3 and S4). WT is the dominant factor in controlling CH$_4$ emission from tropical peatland [24, 39, 45]. At our site, CH$_4$ emission during the dry period ranged from 0.67 ± 0.08 to 1.60 ± 0.47 mgCH$_4$ m$^{-2}$ h$^{-1}$ when WT was low ($–28.1$ cm in July 2019 and $–16.4$ cm in August) and from 3.52 ± 0.83 to 5.96 ± 0.96 mgCH$_4$ m$^{-2}$ h$^{-1}$ at higher WT during the wet period ($–10.7$ cm in October 2018 and $–4.2$ cm in January 2019). We also observed a similar trend along the transect where emission values ranged from 4.33 ± 0.56 to 1.12 ± 0.24 mgCH$_4$ m$^{-2}$ h$^{-1}$ near the drainage canal to the area towards the forest, respectively. These increases in CH$_4$ emissions were mainly attributed to WT, which increased with the seasonal increase in precipitation from August 2018 to January 2019 (figure 2(c)) as well as along the transect. CH$_4$ emission at our study is on the higher side compared to other literature reported from degraded tropical peatlands [24, 38, 46, 47], likely because of the difference in WT depths (table S5) as well as previous studies did not include plant-mediated CH$_4$ transport.
Temperature is known to enhance CH₄ production and emission, where the optimum temperature range is between 25 °C and 40 °C [48, 49]. At our site, CH₄ emissions showed a weak positive correlation with peat soil temperature ($r^2 = 0.08; p < 0.1$) and a contrasting negative correlation with pore-water temperature ($r^2 = -0.41; p < 0.1$) (tables S3 and S4). These trends correspond to the observations from other studies that reported WT as a primary factor affecting CH₄ emission in the tropics instead of temperature change [24, 39, 47, 50, 51].

Pore-water properties were similar across measurements recorded before the February 2019 fire event, whereas, in July 2019, we observed noticeably higher pH, EC, TDS, Salinity, DOC, and pore-water ions concentration (tables 1 and 2). This increase could partly be due to the release of anions and cations from soil organic matter combustion from the fire event [52] as well as due to atmospheric deposition of nutrients from biomass burning in tropical peatland areas [53]. In addition, high Ca²⁺, Mg²⁺ and K⁺ content of ash can increase pH by replacing H⁺ and Al³⁺ ions adsorbed on the negative charge of the soil colloids [52, 54]. This is consistent with our pore-water nutrient analysis, which showed higher K⁺ ($p < 0.05$), Ca²⁺, and Mg²⁺ concentrations in our July 2019 measurements (table 2). We also observed higher PO₄³⁻ ($p < 0.05$) and SO₄²⁻ concentrations, which are known to oxidize CH₄ and inhibit methanogenesis in soils [55, 56] (table 2), hence explaining the lower CH₄ emissions along with the lower WT levels (figures 2–5).

Methanogenesis occurs at an optimum pH level, i.e. near neutral [48, 57]. At our site, the average pH was acidic (3.6 ± 0.2), as reported for other tropical peatland studies [24, 58, 59], and was negatively correlated ($r^2 = -0.67; p < 0.001$, table S4) with CH₄ emissions. However, methanogenesis can occur at low pH with the presence of acid-tolerant methanogens and pH-neutral microsites within the complex peat matrix [60, 61]. Studies from northern regions, however, have reported an inconsistent relationship between pH and CH₄ emissions and further suggested pH as a secondary determinant in CH₄ emission [62]. In addition to pH, we also observed that Salinity had a significant negative correlation ($r^2 = −0.51; p < 0.05$, table S4) with CH₄ emissions, which is consistent with the studies reported from wetlands in the saline environment [63, 64].

### 4.3. Methane emissions from sedge-dominated tropical peatland compared to other land-use types

Our analysis shows that sedge-dominated degraded tropical peatland areas may contribute to 18.0–329.7 kgCH₄ ha⁻¹ yr⁻¹ (118.0 ± 71.4) (figure 7). This number is substantially higher in comparison to values reported from other tropical peatland land use types [65, 66], as well as compared to recent estimates from the intact PSF in Sarawak [67] and Kalimantan [68] (table S5). It is approximately three times the values reported for intact PSF [66, 67], whereas seventeen times higher when compared to emission values reported for similar land-use types, i.e. cropland and shrubland category [66] (table S5). Moreover, such high variation across different land-use presented here could primarily be attributed to the WT level, which is a well known factor in regulating CH₄ emission in tropical peatlands compared to temperature change [24, 39, 47, 50, 51]. In addition to WT, changes in peat soil characteristics due to fire may play a significant role in affecting CH₄ emissions. Specifically, the higher bulk density at the peat surface (supplementary information), the lower DO, EC, PO₄³⁻ and SO₄²⁻ concentrations together with higher pH (tables 1 and 2) may lead to optimum conditions for methanogenesis to occur [24]. Furthermore, differences in vegetation cover (sedges and ferns at our sites) may also drive CH₄ emissions, as changes in vegetation type can affect substrate availability for methanogenesis, as reported by recent studies from tropical peatlands in Panama [69, 70].

We know that the latest (2015) estimate of degraded open tropical peatlands in SE Asia is approximately 1.42 Mha [4]. However, we do not have a clear sedge cover assessment within this area. At our site, the sedge percentage cover varies from 23–31% near the drainage canal to 5–9% away from the drainage canal, i.e. towards the forest. Due to the logistical difficulties in undertaking this kind of research, only a handful of vegetation assessment studies have been pursued in degraded tropical peatlands in SE Asia [71–73]. Our vegetation composition results are similar to other studies [72, 73], where sedge cover ranges from 15% to >60% during different stages of regeneration and restoration of degraded tropical peatland areas. More importantly, sedges are known to out-compete native seedlings during natural regeneration, which may result in these fire-degraded tropical peatland areas becoming sedge-dominated [11, 74]. As our findings confirm that sedge-mediated transport is a significant source of CH₄, it becomes even more important to account for sedge cover change when planning for effective restoration and regeneration strategies of degraded tropical peatlands within a C accounting context.

Recent studies have established that the fire in tropical peatlands originates commonly in deforested areas, where presence of shrubs such as sedges and ferns makes it more prone to recurrent fires [5, 14, 76–78]. Moreover, as prolonged dry-conditions associated with El-Niño are expected to increase in the future [79], these fires can become even more frequent. One of the consequences of repeated fires and drainage on peatland is land subsidence, resulting from combustion and oxidation losses of peat volume, which leads to an increased risk of flooding [15, 75, 80], and eventually impaired vegetation regeneration [17, 72, 81]. Moreover, these already
subsided fire-degraded tropical peatland areas may experience severe flooding where the total flooded area can expand up to five times during the wet period as shown in a 2006 study in Jambi Province, Indonesia, hence expanding the sedge dominated landscapes drastically [17]. Our data also indicates that sedges are more abundant during the wet period and closer to water inundated area near the drainage canal. Further, the chance of an increase in floods can be higher in these areas, which is compounded by a general increase in mean and extreme precipitation projected in SE Asia by 2100 [82]. Within this context, there is a potential for a vegetation shift from fern-dominated to more flood-tolerant sedge-dominated landscapes, where fire-degraded tropical peatland areas may play an even more significant role within the CH\(_4\) budget at the regional level in the near future.

To summarize, we found that the WT level, pore-water properties, and sedge cover significantly correlated with CH\(_4\) emission. As our results are based on observations from one study site with measurements recorded over four field campaigns in one year period, we acknowledge the uncertainties associated with our analysis and its replicability in similar tropical peatland ecosystems. Therefore, we suggest a spatially distributed study across multiple fire-degraded tropical peatland sites over a longer temporal scale to investigate further these relationships as it can be affected by the factors such as seasonality, fire events, and distance from drainage canal.

5. Conclusion

In this study, we present the first measurements of sedge-mediated CH\(_4\) emissions in degraded tropical peatland, which is primarily occupied with sedges and ferns. As the extent of such degraded tropical peatland areas has increased to one-tenth of the total peatland area in SE Asia over the past few decades, their contribution to the regional C emissions is becoming more critical. We found that sedge-mediated transport contributed >70% of the total CH\(_4\) emissions to the atmosphere and was approximately 17 times higher when compared to cropland & shrubland land-use type or up to three times higher than intact PSF in SE Asia. The rate of such emissions strongly correlated with the WT, peat soil water content, number of sedges, distance from the drainage canal, and pore-water properties. We also observed significant differences in pore-water properties from previously burnt and recently burnt plots, significantly affecting CH\(_4\) emissions. As these fire-degraded areas experience frequent flooding, which transforms them into wetlands, it can result in ecological succession from fern-dominated to flood tolerant sedge-dominated landscapes, thereby resulting in higher CH\(_4\) emissions from these areas than previously thought.

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Data availability statement

The data that support the findings of this study are available upon reasonable request from the authors.

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