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Title	Assessing the carbon dioxide balance of a degraded tropical peat swamp forest following multiple fire events of different intensities
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22 Abstract:

23 Tropical peat swamp forest is a unique ecosystem rich in carbon and water, 24 accumulating a huge amount of carbon as peat. However, the huge carbon pool has been threatened by oxidative peat decomposition and fire loss mainly because of deforestation 25 26 and drainage. Fire causes acute carbon dioxide (CO₂) emissions through the combustion 27 of biomass and peat. Also, fire should change the CO_2 balance of postfire ecosystems. 28 Although it is crucial to quantify CO_2 balance even after a fire event to understand the 29 total fire impact, information based on field measurement is lacking. Thus, we had 30 measured eddy CO₂ flux above a repeatedly burned degraded peat forest for about 13 years since 2004. The site was a stable CO₂ source of 147–290 g C m⁻² yr⁻¹ for five years 31 after a stand-replacing fire in 2002. Unexpectedly, a moderate-severity fire in 2009 32 33 changed the site to a CO_2 sink of about $-600 \text{ g C m}^{-2} \text{ yr}^{-1}$. The drastic change would have 34 been caused by a large decrease in the decomposition of plant debris, which had 35 accumulated since the 2002 fire but was burned by the 2009 fire. In contrast, gross primary production (GPP) remained about the same even though vegetation was damaged, 36 37 mainly because year-round wet conditions caused by a La Niña event promoted the regrowth of hygrophilous herbaceous plants and were favorable to their GPP. The site 38 39 also had a low-severity fire and was drained in 2014 but did not return to a CO₂ source. 40 However, the net ecosystem CO₂ uptake after the 2009 fire was insufficient to recover a 41 large amount of fire CO₂ emission. If CO₂ emissions from four fires in 1999, 2002, 2009 42 and 2014 are counted, the site is expected to owe an outstanding CO₂ debt of 25 kg C m⁻ 2 43

45 **1. Introduction**

Tropical peatlands are a huge soil carbon pool of 105 Gt C (Dargie et al., 2017), of 46 47 which 65% have accumulated in insular Southeast Asia, coexisting with peat swamp forest (Page et al., 2011). In recent decades, however, peat swamp forest has experienced 48 49 two major disturbances: land conversion and fire. Land conversion accompanied by deforestation is mainly to develop agricultural land, such as oil palm and acacia 50 51 plantations. In Indonesia and Malaysia, peat swamp forest decreased by 7.3 Mha from 52 1990 to 2015, and consequently plantations and small holders' farmlands occupy more 53 than 50% of tropical peatlands' area (Miettinen et al., 2016). Also, vast deforested area is 54 left unused as secondary forests, shrublands and grasslands (Miettinen et al., 2016). 55 Drainage is common practice in land conversion to lower the groundwater level (GWL), which potentially enhances oxidative peat decomposition and increases soil carbon 56 dioxide (CO₂) efflux (e.g. Hirano et al., 2009; Sundari et al. 2012; Wakhid et al., 2017; 57 58 Itoh et al., 2017).

59 Intact peat forest does not burn easily because of high GWL (Page and Hooijer, 2016). However, peat aridification due to drainage raises the risk of peat fire, especially in the 60 61 dry season because dry peat is flammable. In forests on mineral soil, fire burns 62 aboveground biomass (AGB) and plant debris (flaming). Besides flaming, peat can keep 63 burning belowground (smoldering). Peat fire frequently occurs near drainages and 64 reoccur there though burned peat depth decreases with fire recurrence (Konecny et al., 2016). In Indonesia, wildfires have occurred every year in the dry season (Gaveau et al., 65 66 2014). The fires enlarge in area and severity when an El Niño event occurs, because the 67 event causes drought. Through biomass and peat burning, a large amount of CO₂ is emitted, and smoke produces haze. Large-scale haze covers maritime Southeast Asia and 68

69 causes economic damage and health problems (Page and Hooijer, 2016). In addition, haze 70 diffuses and attenuates solar radiation incident on the ground, which affects gross primary 71 production (GPP) (Kobayashi et al., 2005; Hirano et al., 2012; Stiegler et al., 2019). For 72 instance, in 1997, when one of the strongest El Niño events occurred, the area of 9.7 Mha 73 burned over Indonesia, of which peatlands occupied 15% (Murdiyarso and Adningsih, 2007), and CO₂ of 0.81–2.57 Gt C was emitted through biomass and peat burning (Page 74 75 et al., 2002). In 2015, another strong El Niño year, large-scale fires burned 4.6 Mha over 76 Indonesia, of which peatlands occupied 37%, and emitted 0.24 Gt C of CO₂ (Lohberger 77 et al., 2017), accounting for 16% of the annual global CO₂ emissions through land-use 78 change (Friedlingstein et al., 2019).

79 Fires severely disturb ecosystems and consequently change their carbon balance. Thus, 80 in addition to estimating the large amount of pyrogenic CO₂ emission, it is essential to 81 quantify the carbon balance of postfire ecosystems to assess how much carbon is lost in 82 total by a fire event. Fire disturbances in peatlands include loss or degradation of 83 vegetation, change in the amount of plant debris, loss of surface peat, change in peat 84 properties and change in soil microbial community. The loss or degradation of vegetation 85 decreases GPP and plant respiration, increases soil temperature (Hirano et al., 2014) and 86 decreases evapotranspiration (Hirano et al., 2015) because of the decrease in the leaf area 87 index (LAI). The amount of plant debris increases or decreases depending on fire severity 88 and frequency (Hanula et al., 2012). When plant debris decreases, heterotrophic 89 respiration can decrease, and vice versa. Surface peat is burned and lost especially by 90 smoldering. As a result, recalcitrant deeper peat is exposed (Lupascu et al., 2020), and the 91 peat surface subsides, which relatively raises GWL (Konecny et al., 2016) and might 92 increase CH₄ emissions (Hirano et al., 2009). Thus, the peat loss potentially suppresses

93 oxidative peat decomposition. Fire burning also changes peat physicochemical properties, 94 including an increase in pH, bulk density and ash content and decrease in nitrogen and 95 organic carbon (Smith et al., 2001; Dikici and Yilmaz, 2006). The effects of these changes 96 on carbon cycles are complex and depend on fire severity and frequency (Neary et al., 97 1999). Besides burning, fire makes soil organic matter (SOM) more stable by heating, thereby reducing microbial respiration (Flanagan et al., 2020). In addition, burning affects 98 99 soil microbial community structure and can alter carbon-cycling processes (Ward et al., 100 2012).

101 The overall CO₂ balance of postfire ecosystems has been measured by the eddy 102 covariance (EC) and chamber techniques, especially in mid and high latitudes. A 103 chronosequnce study using EC data showed that most forests in North America changed 104 to a large CO₂ source immediately after stand-replacing fires and returned to a CO₂ sink 105 in 10 years (Amiro et al., 2010). Another chronosequnce study using chamber data 106 showed that boreal bogs in Canada returned to a CO₂ sink in 13 years on average after 107 severe fires (Wieder et al., 2009). Severe fire changed forests and bogs to a CO₂ source 108 owing to a large decrease in GPP, whereas ecosystem respiration (RE) showed a small 109 change (Amiro et al., 2010; Dore et al., 2012; Wieder et al., 2009). Following gradual 110 GPP increase with vegetation regrowth after fire, net CO₂ uptake becomes positive 111 (Ueyama et al., 2019). In tropical peatland, only Hirano et al. (2012) reported the CO_2 112 balance of a burned degraded swamp forest. They showed that the degraded forest was a 113 stable CO₂ source during 1–5 years after a stand-replacing fire. However, all the above-114 mentioned results are from stand-replacing or severe fires. Fire effects on ecosystem CO2 115 balance should depend on combustion severity, but related information based on field 116 studies is lacking (Meigs et al., 2009).

117 We started measuring eddy CO_2 flux in a severely burned degraded peat swamp forest in Indonesia in 2004 (Hirano et al., 2012). The site was burned moderately again in 2009. 118 119 Although many of regrowing trees survived, the measurement was forced to be suspended 120 because of fire damage to the power system. We resumed the measurement after 2.5 121 months of suspension and continued it until the end of 2016. The site was lightly burned 122 again and drained in 2014. In this study, we assessed the CO₂ balance of the degraded 123 swamp forest and investigated the effects of multiple fire events of different intensities 124 and drainage as well as El Niño / La Niña events on the CO₂ balance.

In Southeast Asia, it is projected that global warming will increase extreme drought occurrence during this century (Cai et al., 2014; Cai et al., 2015; Cook et al., 2020). In addition, human-induced disturbance on peatland hydrology will increase drought severity in Indonesia (Taufik et al., 2020). These predictions indicate that fire risk will increase in tropical peatlands. Thus, we must understand more about fire effects on tropical peatland CO₂ balance.

131

132 **2. Materials and methods**

The study site (2.34°S, 114.04°E) was in tropical peatland near Palangkaraya, Central Kalimantan, Indonesia, which was called DB in our previous paper (Hirano et al., 2012). The site was in Block C of ex-Mega Rice Project (Page et al., 2009) and used to be a selectively logged peat swamp forest with a peat depth of about 4 m but burnt repeatedly in 1997, 2002, 2009 and 2014, El Niño years. The forest was severely damaged by standreplacing fire in 2002, leaving a few standing dead trees and a lot of coarse woody debris (CWD) on the ground (Fig. 1a). The thickness of burned peat by the 2002 fire was 0.22

¹³³ **2.1 Study site**

m on average in the study site (Hirano et al., 2014). Peat probably burned also in 1997,
but no data were available on the combustion thickness. A large canal was excavated in
1996–1997 in the area and had functioned to drain the site (Page et al., 2009).

144 A 4-m-tall tower was built at about 200 m from the canal in April 2004. Fetches for 145 the west, south, north and east directions were 250, 300, 1000 and 1000 m, respectively. 146 In April 2004, 1.5 years after the 2002 fire, ferns (Stenochlaena, Blechnum and Lygodium 147 spp.) and sedge (*Cyperus*, *Scleria* and *Eleocharis* spp.) plants were sparsely distributed, 148 of which Stenochlaena palustris was dominant. The ground was studded with fire scars 149 with water (Fig. 1b). Fern and sedge plants had grown up to 0.5 m and covered most parts 150 of the ground in June 2005 (Fig. 1c). A few young trees had regenerated, and the canopy 151 height reached 0.8-0.9 m on average in 2009 before the next fire (Fig. 1d). A fire 152 moderately burned the study site in September 2009. Although stand replacement did not 153 occur, some trees died, and others defoliated. The aboveground part of herbaceous plants 154 burnt. A considerable amount of CWD on the ground burned out. The peat surface burned 155 in spots (Fig. 1e). Power cables and ground sensors were severely damaged. After the 156 moderately-severity fire, surviving trees started to foliate under high groundwater level 157 (GWL) conditions, and ferns and sedge plants rapidly regrew and covered most of the 158 ground in December 2009 (Fig. 1f). Sparsely existing young trees, which was dominated 159 by Combretocarpus rotundatus, kept growing with dense understory; the canopy height 160 exceeded 2 m in 2013 (Fig. 1g). In May–June 2014, a small canal with a width of about 161 1.5 m was excavated by the local community at about 100 m north of the tower and 162 lowered GWL (Fig. 2b). The study site was burned again by a light-severity fire in 163 September 2014. Just before the fire, tree density with a diameter at breast height (DBH) > 3 cm was 860 trees ha⁻¹, and aboveground biomass (AGB) was 19.0 t ha⁻¹. The fire 164

165 burned herbaceous plants, litter accumulation on the ground and young trees in spots, 166 though many trees survived from burning (Fig. 1h). In 2015, a strong El Niño year, 167 although some fires occurred in the area, the site did not burn but was covered with dense 168 smoke. The regrowth of herbaceous plants was slower after the 2014 fire than after the 169 2009 fire, probably because of lowered GWL by El Niño drought and canal excavation. 170 The mean annual temperature and precipitation measured at 1.5 m height for 2005– 171 2016 were 26.4 \pm 0.3°C and 2640 \pm 473 mm (mean \pm 1 standard deviation (SD)), 172 respectively. The dry season (monthly precipitation < 100 mm) usually lasts for 3–4

173 months between July and October. Between 2000 and 2016, El Niño events occurred in 1742002-2003, 2004-2005, 2006-2007, 2009-2010, and 2014-2016, whereas La Niño 175 events occurred in 2000-2001, 2005-2006, 2007-2008, 2008-2009, 2010-2012, and 176 according Oceanic Index 2016 to Niño (NOAA; 177 https://origin.cpc.ncep.noaa.gov/products/analysis monitoring/ensostuff/ONI v5.php). 178

179 **2.2 Measurement of flux and environmental factors**

180 Measurement was continuously conducted from April 2004 to December 2016, though 181 flux data were available until October 25, 2016 because of sensor malfunction. Eddy CO₂ 182 flux was measured using a sonic anemometer-thermometer (CSAT3; Campbell Scientific 183 Inc., Logan, UT, USA) and an open-path CO₂/H₂O analyzer (LI7500; Li-Cor Inc.) using 184 the EC technique at 3.0 m height (April 2004 to February 2011), 3.6 m (February 2011 to 185 March 2012), 7.2 m (March 2012 to December 2013) or 13.6 m (after December 2013). 186 The height was raised following tree growth. The flux sensors were separated by 0.2 m. 187 Eddy signals were collected with a data logger (CR1000; Campbell Scientific Inc.) at 10 188 Hz. CO₂ concentration was also measured at heights of 0.4, 0.7, 1.5, 2.7, 5.7 and 13.6 m in rotation with a closed-path CO₂ analyzer (LI820; Li-Cor Inc.) after December 2013.
Sampling height was switched every 30 seconds, and the mean for the last 5 s was
recorded with another datalogger (CR1000; Campbell Scientific Inc.). The two CO₂
analyzers were calibrated every three months using two standard gases.

193 Air temperature and relative humidity were measured using a platinum resistance 194 thermometer and a capacitive hygrometer (HMP45; Vaisala, Helsinki, Finland), 195 respectively, installed in a radiation shield (DTR503A; Vaisala) at 1.5 m height. 196 Additionally, air temperature and relative humidity were measured at 7.2 m from June 197 2012 to December 2013 or at 13.6 m from December 2013 to December 2016. 198 Precipitation was measured with a tipping-bucket rain gauge (TE525; Campbell Scientific 199 Inc.) at 1.5 m. Downward photosynthetic photon flux density (PPFD) was measured at 200 3.3 m (April 2004 to March 2012), 6.8 m (March 2012 to December 2013) or 13.6 m 201 (December 2013 to December 2016) with a quantum sensor (LI-190S; Li-Cor Inc. or 202 PQS1; Kipp & Zonen, Delft, The Netherland). Soil temperature at 5 cm depth was 203 measured at two points with thermocouple thermometers. These sensor signals were 204 collected every 10 s, and their half-hourly means were recorded with dataloggers 205 (CR1000 and CR10X; Campbell Scientific Inc.). GWL was measured half hourly with a 206 water level logger (DL/N; Sensor Technik Sirnach AG, Sirnach, Switzerland or DCX-22 207 VG; Keller AG, Winterthur, Switzerland) at a single point, avoiding fire scars. GWL was 208 relative groundwater level from the ground surface. When the water surface was 209 belowground, GWL was negative.

The measurement was suspended from September 20 to December 4, 2009 because of power problems due to fire damage. For the suspension period, we substituted data measured at the drained forest site (DF) about 600 m away from this site (Hirano et al., 2012) for precipitation and PPFD, which were basically independent of ground conditions. Also, vapor pressure deficit (VPD), GWL and soil temperature, which were used for the gap filling of CO₂ fluxes, were estimated from the DF's data using significant linear relationships (P < 0.001). Linear regression was conducted for the period between September and December 2006 in another El Niño year; R^2 was 0.89 (VPD), 0.95 (GWL) and 0.48 (soil temperature).

219

220 **2.3 Flux calculation and quality control**

221 Half-hourly eddy CO₂ flux was calculated using Flux Calculator software (Ueyama et 222 al., 2012) with the following procedures: (1) the removal of noise spikes (Vickers and 223 Mahrt, 1997); (2) double rotation for tilt angles; (3) correction for high frequency loss 224 (Massman, 2000; Massman, 2001); (4) correction for air density fluctuations (Webb et al., 225 1980). To avoid flow distortion by the tower, data with wind directions within 20° from 226 the north were excluded; 19% of data were excluded in total. Data during rain were also removed. In addition, data beyond the range of mean ± 3 SDs (> 50 or < -50 μ mol m⁻² s⁻ 227 228 ¹) were screened out as outliers. We checked the stationarity for each 30-min run (Foken 229 and Wichura, 1996). We calculated difference between covariance values for the whole 230 30 min and the average of six 5-min covariance values. Data were excluded, when the 231 difference was more than 100% (Foken et al., 2004).

The CO₂ storage change (FS) under the flux measurement height was calculated every three minutes from CO₂ concentrations measured at a height with the open-path analyzer before December 2013 or CO₂ profiles after December 2013, and then their mean and SD were calculated half hourly. The half-hourly mean FS was also quality-controlled using the SD by the same method as in Hirano et al. (2007, 2012). Net ecosystem CO₂ exchange 237 (NEE) was calculated as the sum of the eddy CO₂ flux and FS.

To exclude underestimated nighttime NEE under calm conditions, we examined the dependency of nighttime NEE on friction velocity (u_*) by the same method as in Hirano et al. (2007). As a result, the dependence was found only during the period after July 2014, probably because trees increased in height. Nighttime NEE with $u_* < 0.074$ m s⁻¹ was excluded only after July 2014, which is defined later as Period III. Through all quality control procedures, $45 \pm 19\%$ and $61 \pm 7\%$ (mean ± 1 SD) of NEE data were missed in the daytime and nighttime, respectively, each year.

245

246 **2.4 Gap-filling and partitioning of NEE**

247 Missing NEE data were gap-filled by the lookup table method on a half-hourly basis. Lookup tables were created every two months using GWL and soil temperature for 248 ecosystem respiration (RE), and PPFD and VPD for gross primary production (GPP). 249 First, we extracted quality-controlled NEE data in the nighttime (PPFD $< 5 \mu$ mol m⁻² s⁻¹) 250 as RE, including CO₂ emissions through burning. The RE data were binned equally into 251 252 ten classes according to GWL, and then the data in each class were binned equally into three classes according to soil temperature and averaged. Using the lookup table, RE was 253 estimated from GWL and soil temperature also in the daytime (PPFD \geq 5 µmol m⁻² s⁻¹). 254 255 GPP was calculated as a difference between the measured daytime NEE and estimated 256 daytime RE (= RE – NEE). The GPP was binned equally into ten classes by PPFD and then into three classes by VPD to create a lookup table. Thus, daytime NEE was gap-257 258 filled as a difference of estimated RE and GPP using the look-up tables.

259

260 **2.5 Uncertainty in annual CO₂ fluxes**

261 Considering the influence of the disturbances of fire and canal excavation, we divided 262 the measurement period into three for the following analyses: Period I (before the 2009 263 fire; April 2004 to September 2009), Period II (after the 2009 fire until the canal 264 excavation; December 2009 to June 2014) and Period III (after the canal excavation; July 265 2014 to December 2016). The annual values of RE, GPP and NEE were compared among 266 three periods by one-way ANOVA and Tukey's HSD using a statistical software R 267 (version 3.6.1).

268 Uncertainties due to random errors and gap-filling in annual RE, GPP and NEE were evaluated using the 24-h differencing approach (Hollinger and Richardson, 2005; 269 270 Richardson et al., 2006; Richardson and Hollinger, 2007). Difference (δ) between two 271 flux measurements exactly 24 hours apart was calculated as a random error (noise), if environmental conditions were similar within 75 µmol m⁻² s⁻¹ in PPFD, 3°C in air 272 temperature, 1 m s⁻¹ in wind speed and 2 hPa in VPD. In each of the three periods, δ was 273 274 sorted according to NEE and binned equally into ten classes in positive and negative NEE ranges, respectively, and then SD of δ ($\sigma(\delta) = \sigma(x_1 - x_2)/\sqrt{2}$)) and mean NEE were 275calculated in each bin. The $\sigma(\delta)$, which is the random measurement uncertainty, was 276 277 significantly correlated to mean NEE (Eq. 1) in each period (P < 0.01).

278
$$\sigma(\delta) = a + b|NEE|$$
 (1)

where *a* and *b* are fitting parameters. Using the equation, $\sigma(\delta)$ was estimated from NEE. We conducted Monte Carlo simulations using R (version 3.6.1) to add a noise (δ) to each of half-hourly measured NEE according to a Laplace distribution, which was parameterized by β (= $\sigma/\sqrt{2}$) (Richardson and Hollinger, 2007). Using the data set of noise-added measured NEE, daytime RE and GPP were estimated, and RE, GPP and NEE were gap-filled by the look-up table method described above to calculate their annual values. The Monte Carlo simulation was conducted 100 times, and the SD of the annual
values were calculated as uncertainty due to random errors.

287

288 **2.6 GPP parameters and peat decomposition**

To understand the interannual variation of the relationship between GPP and PPFD, the following non-rectangular hyperbola (Thornley, 1976) was used as a fitting curve for half-hourly measured data in each year.

292
$$GPP = \frac{\alpha \cdot PPFD + GPP_{max} - \sqrt{(\alpha \cdot PPFD + GPP_{max})^2 - 4\alpha \cdot PPFD \cdot \theta \cdot GPP_{max}}}{2\theta}$$

(2)

where α is the initial slope of the fitting curve, GPP_{max} (µmol m⁻² s⁻¹) the photosynthetic capacity at light saturation and θ the degree of curvature. To reduce the number of parameters, θ was set to be 0.8 (Johnson et al., 2010; Raj et al., 2016). The other two parameters (α and GPP_{max}) were determined using R (version 3.6.1).

298 CO₂ emissions through oxidative peat decomposition (PD; μ mol m⁻² s⁻¹) were 299 estimated from GWL (m) using the following equation, which was determined from soil 300 chamber data at this study site (Hirano et al., 2014).

$$301 PD = 1.48 \cdot ln(1.48 - 5.96 \cdot GWL) (3)$$

302

303 **2.7 Vegetation index and fire hotspots**

304The enhanced vegetation index (EVI) in moderate resolution imaging305spectroradiometer (MODIS) data (MOD13Q1 Version 6) was downloaded from Land306ProcessesDistributedActiveArchive307(https://lpdaac.usgs.gov/products/mod13q1v006/). Pixel data covering our site were used

308	as vegetation information, such as a proxy of AGB (Anaya et al., 2009; Huete et al., 2002).
309	The data were composited every 16 days at 250 m spatial resolution. Also, MODIS
310	hotspot data (MODIS Collection 6) acquired from the Fire Information for Resources
311	Management System (https://firms.modaps.eosdis.nasa.gov/) were used to know the fire
312	occurrence.

313

314 **3. Results**

315 **3.1 Time-series variations**

316 Precipitation showed a clear seasonal variation, except in 2008 and 2010, La Niña 317 years (Fig. 2a). Following the seasonal precipitation variation, GWL showed a negative 318 peak at the end of the dry season (Fig. 2b). The lowest monthly mean GWL was -0.92 m 319 in 2006 and -1.38 m in 2015 before and after the canal excavation (Table 1). During the 320 mid-wet season, typically between January and April, groundwater rose aboveground 321 (GWL > 0 m) until 2014. In 2010, however, there was no dry month, and consequently 322 GWL was stable at around 0 m throughout the year. In addition, GWL remained below the peat surface (< -0.17 m monthly) even in the rainy season after canal excavation in 323 324 2014. In comparison between 2011 and 2016, which had similar annual precipitation 325 (Table 1), annual mean GWL was lower by 0.38 m in 2016 after drainage. Fire occurred 326 in the dry season when GWL lowered. Many hotspots were detected in 2002, 2009, 2014, 327 and 2015, El Niño years (Fig. 2g). The number of hotspots within a 5-km radius from the 328 tower were 442, 262, 117 and 80, respectively, in 2002, 2009, 2014 and 2015; the shortest distances from the tower were 0.15 (2002), 0.19 (2009), 0.26 (2014) and 1.23 km (2015). 329 330 In 2002, EVI drastically decreased from 0.61 to 0.12 (Fig. 2f). EVI also dropped to 0.28 by the 2009 fire and had gradually recovered toward the pre-fire level of 0.51. In contrast, 331

fire decrease in EVI was temporary for about 10 months in 2014. PPFD decreased in the dry seasons of 2002, 2006, 2009, 2014 and 2015 because of dense smoke emitted from fires (Fig. 2c). Daytime VPD increased in the dry season every year except in 2008 and 2010, La Niña years (Fig. 2d). The VPD peak became high in El Niño years, especially in 2009, 2014 and 2015.

Before the 2009 fire, RE was almost stable with small monthly fluctuations, whereas 337 338 GPP had gradually increased following vegetation regrowth (Figs. 2e and 2f). During the 339 period, NEE was positive except for a few months. In December 2009, 2.5 months after 340 the fire, RE decreased to less than half of the pre-fire level. By contraries, there was little 341 change in GPP even after the fire. As a result, NEE drastically changed from positive to 342 negative. Both RE and GPP were almost stable in 2010, and then started increasing 343 following gradual EVI increase (Fig. 2f). During the fire in 2014, RE showed a sharp peak because of a large amount of CO₂ emission through burning in the study site (Fig. 344 1h). In contrast, GPP suddenly decreased by 2.8 g C m⁻² d⁻¹ between August and 345 346 November (Fig. 2e). These opposite changes resulted in a large NEE peak. After the fire, RE rapidly decreased to less than the pre-fire level, whereas GPP slightly recovered, 347 resulting in a CO₂ neutral (NEE \cong 0 g C m⁻² yr⁻¹) in 2015. In 2016, NEE returned to be 348 349 negative (Table 1).

350

351 **3.2 Environmental responses of RE and GPP**

Generally, RE decreased with increasing GWL in all periods (Fig. 3a). In Period I, RE showed a decreasing tendency only when GWL was positive. In contrast, RE decreased with increasing GWL over the whole GWL range in Period II, though the range was narrower than those of the other periods. If excluding the spiky RE during the fire in

September and October 2014 (Fig. 2e), RE showed a clear negative relationship with 356 357 GWL in Period III. To examine more about RE response to GWL, we estimated the sum 358 of autotrophic respiration and the decomposition of CWD and litter accumulation on the ground by subtracting estimated PD (Eq. 2) from the measured RE (RE - PD). Positive 359 360 relationships were found between RE-PD and GWL in Periods I and III, if spiky fire data were excluded (Fig. 3b); sensitivity to GWL was higher in Period I. RE was almost stable 361 until 2009 (Period I) in high GWL conditions (GWL \geq -0.2 m) (Fig. 4). After the fire, RE 362 363 at high GWL decreased by half in 2010, then had increased until 2014.

364 GPP_{max} increased from 2004 to 2005, and then became almost stable until 2009 (Fig. 365 5). After the 2009 fire, GPP_{max} kept increasing until 2012 and decreased in 2015. To examine the influence of VPD, light-saturated GPP at PPFD $\geq 1000 \ \mu mol \ m^{-2} \ s^{-1}$ 366 (GPP₁₀₀₀) was plotted against VPD for the three periods under different GWL conditions 367 (Fig. 6). In Period I, GPP₁₀₀₀ decreased gently but linearly with increasing VPD regardless 368 369 of GWL conditions. In Periods II and III, however, GPP₁₀₀₀ started decreasing at VPD of about 15 hPa. Larger data scattering at GWL < -0.4 m in Period II and GWL ≥ -0.1 m in 370 Period III is due to smaller data size. The relationship of GPP₁₀₀₀ with VPD was 371 372 independent of GWL conditions.

373

374 **3.3 Annual NEE, RE and GPP**

Annual NEE was positive over 150 g C m⁻² yr⁻¹ for consecutive five years until 2009 (Table 1). After 2010, annual NEE drastically decreased to less than -570 g C m⁻² yr⁻¹ mainly because of the large decrease in RE. The fire halved annual RE from 1488 to 715 g C m⁻² yr⁻¹, whereas annual GPP decreased only by 49 g C m⁻² yr⁻¹ from 2008 to 2010. The decrease in RE–PD (718 g C m⁻² yr⁻¹) contributed much more to the RE decrease than PD decrease (55 g C m⁻² yr⁻¹). After the fire, NEE continued to increase negatively until 2013 because GPP increased more rapidly than RE; increases between 2010 and 2013 were 628 and 735 g C m⁻² yr⁻¹, respectively, in RE and GPP. In 2014, GWL was clearly decreased by canal excavation and less precipitation, and fire occurred in the study site. As a result, NEE changed to almost zero (-28 g C m⁻² yr⁻¹) because of a large increase in RE and a slight decrease in GPP. The large RE increase was attributable to fire CO₂ emissions. The neutral NEE continued until 2015.

To examine the large change of CO₂ fluxes, annual values were averaged in the three periods: 2005–2008, 2010–2013 and 2015–2016 (Table 2). Their interannual variations (1 SD) in each period were greater than uncertainties due to random errors (Table 1). Although a significant difference was not found in GPP (Table 2), RE and NEE were significantly smaller or more negative in 2010–2013 (P < 0.05).

392 We have already reported the CO_2 balance of this site between 2004 and 2009 (Hirano 393 et al., 2012), in which annual periods began in early June. Using the same annual periods, 394 annual CO₂ fluxes were calculated and compared for five years. As a result, annual NEE of this study $(277 \pm 113 \text{ g C m}^{-2} \text{ yr}^{-1})$ was smaller than that of Hirano et al. (2012) by 185 395 g C m⁻² yr⁻¹ on average. This difference was mainly due to RE difference of 233 g C m⁻² 396 yr⁻¹, which was mainly caused by data screening in the nighttime. In Hirano et al. (2012), 397 negative data or data > 100 μ mol m⁻² s⁻¹ were excluded in the nighttime at the beginning 398 of quality control procedures. In contrast, screening criteria were between -50 and 50 399 µmol m⁻² s⁻¹ both in the daytime and nighttime in this study. The biased screening 400 401 overestimated RE in Hirano et al. (2012). Thus, we can say that this study is fairer in 402 quality control and more reliable.

404 **4. Discussion**

405 **4.1 Ecosystem respiration**

406 The RE drastically decreased after the 2009 fire (Fig. 2e). Annual RE was smaller by 773 g C m⁻² yr⁻¹ in 2010 than in 2008 (Table 1). In both years, La Niña events occurred, 407 408 and GWL was high at around the ground surface throughout the year (Fig. 2b). As a result, the difference in the annual PD was small (55 g C m⁻² yr⁻¹). Thus, the large difference in 409 410 RE was due to decreases in autotrophic respiration and other heterotrophic respiration 411 from CWD and litter accumulation on the ground, which were equivalent to RE-PD. The 412 2009 fire burned most of litter and CWD produced by the previous 2002 fire (Fig. 1e), 413 leaving some charred CWD. Charred CWD is slow in decomposition (Deluca and Aplet, 414 2008; Donato et al., 2009). The RE-PD showed a large decrease after the 2009 fire at high GWL (Fig. 3b). These facts indicate that CO₂ efflux through the decomposition of 415 416 CWD and litter accumulation sharply decreased.

417 Smoldering occurred only at some spots during the 2009 fire, but the peat surface 418 burned along with litter accumulation. Such combustion processes replace labile carbon 419 compounds with more recalcitrant charcoal (Milner, 2013). Even if peat does not burn, 420 fire can thermally alter the chemistry of SOM and increase stable soil carbon, which 421 potentially lowers the peat decomposition (Flanagan et al., 2020). Meanwhile, peat 422 combustion increases mineral nutrient, pH and bulk density (Dikici and Yilmaz, 2006; 423 Smith et al., 2001). Higher pH potentially enhances soil microbial activity and 424 consequently increases microbial respiration (Moilanen et al., 2012). The effects of theses pyrogenic changes in peat properties depend on fire severity and frequency (Neary et al., 425 426 1999). Here, the decreasing effects possibly surpassed the increasing effects on carbon 427 emissions.

428 After 2010, RE had gradually increased until the next fire in 2014 (Fig. 2e). During 429 this period, the vegetation recovered with litter accumulation, as shown by EVI increase 430 (Fig. 2f). PD was relatively stable under continuous higher GWL conditions (Table 1, Fig. 431 2b). Thus, the RE increase was attributable to the increase in autotrophic respiration and 432 litter decomposition. GWL lowered to a large extent after 2014 because of canal 433 excavation and El Niño drought (Fig. 2b). Even in the wet season, GWL remained belowground. As a result, PD increased rapidly up to 844 g C m⁻² yr⁻¹ in 2015 (Table 1). 434 435 Eq. 3 to estimate PD was parameterized using the data measured in 2004–2006, in which 436 GWL was higher than -0.9 m (Hirano et al., 2014). Thus, PD in 2015 with the minimum 437 daily GWL of -1.53 m was extrapolated, and the annul PD might be overestimated (Itoh 438 et al., 2017).

439

440 **4.2 Gross primary production**

The annual GPP decreased only by 49 g C m⁻² yr⁻¹ between 2008 and 2010 despite 441 442 30% decrease in EVI (Table 1). After 2010, GPP continued to increase until the 2014 fire 443 (Fig. 2e) with the rapid regrowth of fern and sedge plants dominated by Stenochlaen 444 *palustris*, which is typically growing in permanently damp open places (Chambers, 2013). The annual GPP in 2013 was 2027 g C m⁻² yr⁻¹, which accounts for about 60% of that of 445 a nearby peat swamp forest (UF site) (Hirano et al., 2012). The rapid regrowth was most 446 447 probably due to limited GWL lowering (Fig. 2b). In 2010 immediately after the fire, GWL 448 remained high throughout the year because of a La Niña event; daily mean GWL lowered 449 only to -0.12 m. Thus, the hygrophilous fern and sedge plants could steadily recover with 450 little water stress under sparse overstory. The leaf color usually gets lighter in the dry 451 season because of desiccation, but little color change was found in 2010–2013. In addition, 452 wood and plant ashes would have promoted vegetation regrowth by increasing plant 453 available nutrients (Neary et al., 1999). Also, higher PPFD and lower daytime VPD 454 increased GPP and enhanced plant growth. Light-saturated GPP showed a decreasing 455 tendency with VPD when VPD was higher than 15 hPa (Fig. 6). Such a dry condition 456 occurred less frequently in 2010–2013 (Table 1). A rapid increase in GPP_{max} during this 457 period (Fig. 5) reflects the rapid vegetation regrowth and lower VPD, which was 458 favorable to GPP.

After 2014, canal excavation and an El Niño event dried the site, which lowered GWL and raised VPD. In addition, aboveground vegetation was patchily damaged by the 2014 fire, and PPFD was attenuated sharply in 2015 by dense smoke emitted from surrounding fires. As a result, both the annual GPP and GPP_{max} temporarily decreased in 2015, and then recovered in 2016 with the disappearance of the El Niño event and vegetation recovery (Table 1). The canal effect on GPP by GWL lowering was unclear because El Niño drought and fire occurred concurrently.

466

467 **4.3 Fire emissions**

468 In 2014, large positive CO₂ flux had been measured for about 40 days in September 469 and October (Fig. 2e), when fire occurred in the study site. The large CO₂ efflux should have included fire CO₂ emissions. The fire emissions were estimated to be 308 g C m⁻² 470 471 as a difference between measured RE and estimated RE using look-up tables created 472 before and after the fire. AGB immediately before the fire was 19.0 t ha⁻¹. From an emission factor (EF) of 0.45, which was an average for tropical forest (Akagi et al., 2011), 473 the fire emission of 19.0 t ha⁻¹ corresponds to 36% combustion of AGB (Fig. 1h), if CWD 474 475 and litter did not burn. Through the fire, EVI decreased by about 40% (Fig. 2f).

476 Fire emissions were not measured in 2009, since the flux measurement was suspended. 477 We presumed that AGB before fire was almost the same in 2009 and 2014, because EVI 478 was at the same level (Fig. 2f). We also presumed that AGB burned by 50% through the 479 2009 fire, because EVI decreased by about 50%. Toriyama et al. (2014) conducted a forest survey in the same area and reported that 86 t ha⁻¹ of deadwood remained on average in 480 481 regenerating forests after the 1997 fire. Thus, we presumed that the same amount of CWD 482 remained in the study site and all the CWD burnt out in 2009. Using the EF of 0.45, fire emission was estimated to be 4.30 kg C m⁻² (= $(19 \times 0.5 + 86) \times 0.45$). 483

484 We also estimated fire emissions by fires in 1997 and 2002. In this area, AGB of peat swamp forest was estimated to be 169 t ha⁻¹ on average (Hayashi et al., 2015). According 485 to Toriyama et al. (2014), deadwood in mature forests accounts for 15% of AGB and 86 486 487 t ha⁻¹ of deadwood remained after the 1997 fire. On the presumption that all the fuel mass except the 86 t ha⁻¹ of deadwood was burned by the two fires, fire emission was calculated 488 489 to be 4.88 kg C m⁻² (= $(169 + 169 \times 0.15 - 86) \times 0.45$) using the same EF, though some 490 AGB was left unburned as CWD. Thus, the calculation should be overestimated, but it 491 can be compensated with the underestimation of the 2009 fire emissions. In addition, peat 492 was burned by 22 cm by the 2002 fire. The bulk density and carbon content of shallow peat (0–25 cm) were 173 kg m⁻³ and 53.2% on average, respectively, in the study site 493 (Itoh et al., 2017). Although the peat properties were measured in 2014 after the 2009 fire, 494 495 these apply probably to the peat of 2002. Using another EF of 0.46 determined from Indonesian peat with carbon content of 54.7% (Christian et al., 2003), CO₂ emission 496 through peat combustion was estimated to be 17.5 kg C m⁻² (= $0.22 \times 173 \times 0.46$). The 497 sum (22.4 kg C m⁻²) of the two emissions from AGB and peat was total fire emissions. 498 The emissions accounted for about 70% of mean fire emissions from peatlands burned in 499

500 1997 in Central Kalimantan, Indonesia (Page et al., 2002).

- 501

502 **4.4 Carbon dioxide balance**

503 This study site was a constant CO₂ source until 2009 after a stand-replacing fire with 504 smoldering in 2002. Unburned forest and wetland generally change from a CO₂ sink to a 505 source by severe fire owing to the large decrease in GPP (Amiro et al., 2010; Dore et al., 506 2012; Ueyama et al., 2019; Wieder et al., 2009). In contrast, a moderate-severity fire in 507 2009 changed this site from a CO₂ source to a net annual CO₂ sink because of a large RE decrease with little GPP decrease (Table 1). The annual NEE of -615 g C m⁻² yr⁻¹ in 2010-508 2013 (Table 2) was much more negative than that (-403 g C m⁻² yr⁻¹) of tropical humid 509 510 evergreen forest (Luyssaert et al., 2007).

511 A complete carbon balance assessment of tropical peatlands with high GWL requires 512 accounting for methane (CH₄) emissions. Hirano et al. (2009) measured CH₄ flux on the 513 peat surface in the same area as this study site between 2004 and 2006 by the chamber 514 method and showed a sigmoid relationship between CH₄ flux and GWL. Using the equation on the assumption that parameters were unchanged even after fires and drainage, 515 516 we estimated CH₄ flux from GWL throughout the study period. As a result, annual CH₄ emissions were estimated to be 1.68 ± 0.30 , 1.90 ± 0.36 and 0.14 ± 0.19 g CH₄ m⁻² yr⁻¹ 517 518 on average, respectively, in 2005-2008, 2010-2013 and 2015-2016, which can be converted into 47.6, 53.2 and 3.9 g CO_{2eq} m⁻² yr⁻¹, respectively, using a global warming 519 520 potential (GWP) of 28 over a timescale of 100 years (IPCC, 2013). The result indicates that CH₄ emissions did not significantly change after the 2009 fire but significantly 521 decreased (P < 0.01 according to Tukey's HSD) after the canal excavation in Period III 522 523 because of low GWL.

524 Fluvial organic carbon flux is another essential component in the carbon balance of 525 peat ecosystems. Moore et al. (2013) estimated annual fluxes of dissolved organic carbon 526 (DOC) and particulate organic carbon (POC) as the product of their concentrations and 527 discharge in canals in intact peat swamp forest and drained disturbed peat swamp forest 528 in this study area. They reported that the annual total organic carbon (DOC + POC) flux depended mainly on annual discharge. Annual total organic carbon flux was larger in the 529 530 drained forest because of larger annual discharge, whereas mean concentrations were 531 similar in the intact and drained forests. In our site since canal excavation lowered GWL 532 in 2014 (Fig. 2), discharge should have increased, and consequently fluvial organic 533 carbon flux increased temporarily in this transitional period. Alternatively, latent heat flux 534 normalized by net radiation did not change between Period II and III (Ohkubo et al., under 535 review), indicating that evapotranspiration capacities were almost the same in both periods. Thus, if GWL keeps lowering after drainage, fluvial carbon loss would increase 536 537 owing to discharge increase. However, if GWL remains stable even at a low level, no 538 more increase in fluvial carbon flux would occur.

Cumulative CO₂ emissions, including fire emissions, had been decreasing after the 539 540 2009 fire until canal excavation and fire in 2014 because of the large negative NEE (Fig. 541 7). However, cumulative CO_2 were not compensated in 13 years; it was still positive at about 3 kg C m⁻² at the end of 2016. Although the moderate-severity fire significantly 542 543 changed the annual CO₂ balance of the repeatedly burned peatland, the change was 544 insufficient to recover the acute fire emissions in this period. In addition, if CO₂ emissions from fires in 1999 and 2002 (22.4 kg C m⁻²) is added, cumulative CO₂ emissions increase 545 up to 25.4 kg C m⁻² at the end of 2016. If annual NEE (-615 g C m⁻² yr⁻¹) in 2010–2013 546 (Table 2) continues, the cumulative CO₂ emissions can be compensated in about 40 years, 547

but is this possible? The site has already been drained since 2014. Lowered GWL potentially enhances oxidative peat decomposition and increases fire risk (Konecny et al., 2016). Thus, it should be impossible for the site to recover the total CO_2 emitted by fires. This site was converted to an oil palm plantation soon after the flux monitoring was closed. Although fire risk is expected to be low in plantations, oil palm plantations established on peat have a potential to be a large CO_2 source up to 1 kg C m⁻² yr⁻¹ because of insufficient peat compaction during land preparation (Kiew et al., 2020).

555 The study area was heavily degraded by deforestation and drainage through Mega Rice Project (MRP) (Page et al. 2009). After MRP was terminated at the end of the 1990s, the 556 557 area has been abandoned and burned repeatedly. As a result, in Block B (4490 km²) of 558 ex-MRP, non-woody vegetation area dominated by ferns like this study site occupied 36% 559 in 2005 (Hoscilo et al., 2011). Thus, this study can contribute to quantifying the CO₂ 560 balance of degraded peatland on a regional scale. However, our result might not apply to 561 other degraded sites because postfire CO₂ balance depends on various conditions, such as 562 fire severity, disturbance histories and the hydrological environment. The drastic change of CO₂ balance was probably caused by an appropriate combination of a moderately-563 564 severity fire and consecutive year-round high GWL due to a La Niña event, which was 565 favorable to the rapid regrowth of herbaceous plants and their GPP. More field data 566 measured in various conditions are necessary in collaboration with a modeling approach 567 to robustly assess the CO₂ balance of degraded peat ecosystems.

568

569 **5. Conclusions**

570 The study site burnt two times in 2009 and 2014 and was drained by a canal excavated 571 in 2014 during the study period of about 13 years. Although fire damages were caused by the El Niño drought, the second fire was probably triggered by drainage. In addition, the smoke emitted by nearby fire attenuated PPFD considerably. On the other hand, the rainy weather caused by La Niño events kept GWL high around the ground surface throughout the year. These human and natural disturbances affected CO₂ fluxes compositely.

576 The large decrease in RE after the 2009 fire was probably due to a large decrease in heterotrophic respiration from CWD and litter accumulation. A La Niña event 577 578 immediately after the fire produced a favorable wet condition for herbaceous plants' 579 regrowth and prevented PPFD attenuation due to smoke emissions from fire, which 580 increased GPP. As a result, the degraded peat swamp forest changed from a CO₂ source 581 to a sink. In 2014–2015, unfavorable events of canal construction, El Niño drought and 582 fire occurred concurrently. Consequently, GWL lowered, VPD increased, PPFD 583 decreased, and vegetation was patchily damaged. These events changed NEE, but the 584 study site did not return to a CO₂ source. However, the net ecosystem CO₂ uptake was 585 quite insufficient to recover a large amount of fire CO₂ emissions.

586

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791 Figure legends

Fig. 1 Time-series photos of the study site: (a) November 2002, (b) April 2004, (c) June
2005, (d) September 2009 before fire, (e) October 2009 after fire, (f) December
2009, (g) December 2013 and (h) September 2014.

- 795 Fig. 2 Monthly variations in (a) precipitation, (b) GWL, (c) PPFD, (d) daytime (0900-796 1500) VPD and (e) CO₂ fluxes between 2000 and 2016, along with (f) EVI at 16-797 days intervals and (g) distance of hotspots from the flux tower within a circle of 5 798 km radius. EVI data were classified into two ranks according to reliability: useful 799 or better (solid circles) and others (open circles). The red line denotes a moving 800 average of seven consecutive solid circles. No data were available before March 801 2004 at the study site (grey area). As for precipitation and PPFD, data measured at 802 a nearby forest site (DF) were shown for reference before April 2002. Dashed 803 vertical lines denote the boundaries between three periods (refer to the text).
- 804 Fig. 3 Relationships of (a) measured RE (nighttime NEE) with GWL and (b) difference 805 between RE and oxidative peat decomposition (PD) (measured nighttime RE estimated PD from GWL) with GWL in Period I (April 2004 to September 2009), 806 807 Period II (December 2009 to June 2014) and Period III (July 2014 to December 808 2016). Data in each period were binned equally into 15 classes according to GWL 809 and averaged. Data except during fire in Period III are shown as grey circles. Error 810 bars denote standard errors. Regression lines are drawn for original data if the 811 regression is significant (P < 0.05).
- Fig. 4 Interannual variation in measured RE (nighttime NEE) in high GWL conditions (≥
 -0.2 m) between 2004 and 2016. Error bars denote standard errors.
- Fig. 5 Interannual variation in GPP_{max} between 2004 and 2016. Error bars denote standard

- 815 errors.
- Fig. 6 Relationships of light-saturated GPP at PPFD ≥ 1000 µmol m⁻² s⁻¹ (GPP₁₀₀₀) with
 VPD in three GWL conditions in Period I (April 2004 to September 2009, a–c),
 Period II (December 2009 to June 2014, d–f) and Period III (July 2014 to December
 2016, g–i). Data in each panel were binned equally into 15 classes according to
 VPD and averaged. Error bars denote standard errors.
 Fig. 7 Cumulative CO₂ emissions including fire emissions from May 2004 to December
 2016. Grey lines denote the events of fires and canal excavation.

(a) November 2002



(c) June 2005



(e) October 2009



(g) December 2013



(b) April 2004



(d) September 2009



(f) December 2009



(h) September 2014





829 Fig. 3a















843 Fig. 7



845 **Tables**

Year	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016
NEE (g C m ⁻² yr ⁻¹) ^a	290	290	231	147	163	-577	-570	-629	-684	-28	1	-381
	(17)	(32)	(27)	(31)	(42)	(9)	(16)	(14)	(14)	(34)	(18)	(21)
RE (g C m ⁻² yr ⁻¹) ^a	$\frac{1487}{(20)}$	(30)	1692	1488	1544	(13)	(23)	(23)	1343	1870	1634 (31)	1428
PD (g C m ⁻² yr ⁻¹)	333	466	192	256	464	201	259	333	301	483	844	(32) 749
$RE-PD (g C m^{-2} yr^{-1})$	1155	1062	1500	1232	1080	514	698	975	1042	1386	790	679
GPP ($g C m^{-2} vr^{-1}$) ^a	1197	1238	1461	1341	1380	1292	1527	1937	2027	1898	1633	1809
	(13)	(23)	(26)	(35)	(35)	(10)	(18)	(19)	(16)	(25)	(19)	(22)
GwL (m)	-0.08	-0.23	0.00	-0.03	-0.23	0.01	-0.04	-0.08	-0.06	-0.22	-0.39	-0.43
Minimum monthly GWL (m)	-0.44	-0.92	-0.28	-0.32	-0.93	-0.03	-0.30	-0.33	-0.36	-0.74	-1.38	-0.85
Precipitation (mm yr ⁻¹)	2620	1977	2555	2603	2239	3750	3021	2460	2776	2220	2388	3077
PPFD (kmol m ⁻² yr ⁻¹)	13.0	12.5	13.8	13.9	15.1	14.7	15.6	14.9	14.4	12.8	11.9	13.4
Daytime VPD (hPa) ^b	15.6	17.2	16.8	16.2	17.5	13.9	15.3	15.5	16.0	18.1	18.8	16.0
Ratio of daytime VPD > 15 hPa ^b	0.56	0.61	0.64	0.59	0.65	0.43	0.53	0.55	0.57	0.66	0.72	0.59
EVI	0.414	0.427	0.422	0.429	0.411	0.310	0.361	0.412	0.431	0.386	0.385	0.444

846	Table 1. Annual val	ues of CO ₂ fluxes an	d environmental factors.
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^a Uncertainty due to random errors is shown in parentheses.

^b Data in the daytime (0900–1500).

- 849 Table 2.
- 850 Mean values (± 1 standard deviation) of CO₂ fluxes and environmental factors in the three periods of 2005–2008, 2010–2013 and 2015–
- 851 2016. The *P* values of one-way ANOVA were also shown. Different letters in the same row denote significant difference among periods
- 852 (P < 0.05) according to Tukey's HSD.

Year	2005–2008	2010–2013	2015–2016	ANOVA (P)
NEE (g C m ⁻² yr ⁻¹)	240±68a	-615±53c	-190±270b	<0.001
$RE (g C m^{-2} yr^{-1})$	1549±97a	1081±300b	1531±146ab	0.035
$PD (g C m^{-2} yr^{-1})$	312±118a	274±57a	794±67b	< 0.001
$RE-PD (g C m^{-2} yr^{-1})$	1237±188a	807±246b	735±78ab	0.031
$GPP (g C m^{-2} yr^{-1})$	1309±118a	1696±346a	1721±124a	0.106
GWL (m)	-0.09±0.11a	-0.04±0.04a	-0.51±0.11b	0.001
Minimum monthly GWL (m)	-0.49±0.29ab	-0.26±0.15b	-1.12±0.37a	0.019
Precipitation (mm yr ⁻¹)	2339±309a	3002±549a	2733±487a	0.276
PPFD (kmol m ⁻² yr ⁻¹)	13.3±0.67ab	14.9±0.51b	12.7±01.06a	0.011
Daytime VPD (hPa) ^a	16.5±0.70a	15.2±0.90a	17.4±1.98a	0.100
Daytime ratio of VPD>15 hPa ^a	0.60±0.03a	0.52±0.06a	0.66±0.09a	0.067

853 ^a Data in the daytime (0900–1500).