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Title: Carbon dioxide emissions through oxidative peat decomposition on a burnt tropical peatland

Running title: Tropical peat decomposition on a burnt area

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Abstract

In Southeast Asia, a huge amount of peat has accumulated under swamp forests over millennia. Fires have been widely used for land clearing after timber extraction, thus land conversion and land management with logging and drainage are strongly associated with fire activity. During recent El Niño years, tropical peatlands have been severely fire-affected and peatland fires enlarged. To investigate the impact of peat fires on the regional and global carbon balances, it is crucial to assess not only direct carbon emissions through peat combustion but also oxidative peat decomposition after fires. However, there is little information on the carbon dynamics of tropical peat damaged by fires. Therefore, we continuously measured soil CO_2 efflux (RP) through oxidative peat decomposition using six automated chambers on a burnt peat area, from which about 0.7 m of the upper peat had been lost during two fires, in Central Kalimantan, Indonesia. The RP showed a clear seasonal variation with higher values in the dry season. The RP increased logarithmically as groundwater level (GWL) lowered. Temperature sensitivity or Q_{10} of RP decreased as GWL lowered, mainly because the vertical distribution of RP would shift downward with the expansion of an unsaturated soil zone. Although soil temperature at the burnt open area was higher than that in a near peat swamp forest, model simulation suggests that the effect of temperature rise on RP is small. Annual gap-filled RP was 382±82 (the mean ± one standard deviation of six chambers) and 362±74 gC m^{-2} y^{-1} in 2004-2005 and 2005-2006 years, respectively. Simulated RP showed a significant negative relationship with GWL on an annual basis, which suggests that every GWL lowering by 0.1 m causes additional RP of 89 gC m^{-2} y^{-1}. The RP accounted for 21−24% of ecosystem respiration on an annual basis.

Introduction

In Southeast Asia, peat has accumulated under swamp forests over millennia (Dommain *et al.*, 2011; Page *et al.*, 2004). In this region, peatlands cover 2.48×10^5 km^2, which accounts for 56% of the whole tropical peatland area, and store up to 68.5 Pg of soil carbon, which accounts for 77% of tropical peat carbon pool (Page *et al.*, 2011). The peat swamp forest ecosystem, however, has been devastated by logging and land development involving deforestation and drainage since the 1970's (Sorensen, 1993), and the degradation of these peat ecosystems has accelerated during the last two decades, resulting in a large decrease in the proportion of forest cover on peatlands, from 77% to 36% in Malaysia and Indonesia (Miettinen *et al.*, 2012b). As a result, large areas of peat swamp forest have been converted into secondary forests, plantations and open shrub and fern lands. The proportion of open land increased from 7% to 15% over the two decades (Miettinen *et al.*, 2012b). Deforestation raises soil temperature (Sano *et al.*, 2010), and drainage directly lowers groundwater level (GWL). Land conversion with deforestation and drainage accelerates oxidative peat decomposition because of temperature rise and increased aeration (Couwenberg *et al.*, 2009; Hooijer *et al.*, 2012). Plantations of oil palm and Acacia covered 20 % of peatlands in Malaysia and Indonesia in 2010, and annual oxidative carbon decomposition from these plantation areas was estimated at 63-85 TgC y^{-1} (Miettinen *et al.*, 2012a).

Fires have been widely used in Southeast Asia for land clearing after timber extraction for making new croplands and plantations (Murdiyarso & Adiningsih, 2007). Thus land conversion and land management are strongly associated with fire activity. In addition, once peat swamp forests are degraded by logging, drainage and/or fires, the risk of fires rapidly increases because such disturbances typically dry the peat surface and leave much plant debris, which are flammable (Cochrane, 2003; Miettinen *et al.*, 2012b; Page *et al.*, 2002). Langner & Siegert (2009) analyzed hotspots detected by satellites in Borneo Island over 10 years from

1997. They reported that 3.8×10^4 km^2 of peat swamp forests were affected by fires over the 10 year period and that peat swamp forests were most severely fire-affected in Kalimantan, Indonesian Borneo, with a fire incidence of 4.6% y^{-1} on an area basis. The fire-affected area of peat swamp forest was more than three times larger in El Niño years than in non-El Niño years (Langner & Siegert (2009). In the former Mega Rice Project (MRP) area in Central Kalimantan, which was heavily devastated by drainage and deforestation, large-scale fires occurred in the El Niño years of 1997, 2002, 2006 and 2009. Burnt peat depth was reported to be 0.51 (Page *et al.*, 2002), 0.27 (Couwenberg *et al.*, 2009; Usup *et al.*, 2004) and 0.33 m (Ballhorn *et al.*, 2009) on average in 1997, 2002 and 2006, respectively. Large-scale peat fires seriously damage peat swamp ecosystems and result in huge amounts of carbon emissions to the atmosphere through peat combustion (Ballhorn *et al.*, 2009; Page *et al.*, 2002). While direct carbon emissions through peat combustion must be quantified to investigate the impact of peat fires on the regional and global carbon balances, it is also crucial to assess oxidative decomposition of peat after fires for a better holistic understanding of ecosystem-scale fire impact. However, there is little information on the carbon dynamics of tropical peat damaged by fires.

There are several studies on the oxidative decomposition of tropical peat and its consequent CO_2 emissions using subsidence rate combined with data on peat bulk density and carbon content (Couwenberg *et al.*, 2009; Hooijer *et al.*, 2010; Hooijer *et al.*, 2012). These show simple relationships between peat decomposition and GWL. Although the approach is relatively applicable, large uncertainties remain in the determination of the oxidative component of subsidence (Murdiyarso *et al.*, 2010). The direct measurement of CO_2 emissions through oxidative peat decomposition is essential to obtain accurate data. However, there has only been one study to directly measure CO_2 emissions, i.e. peat decomposition, using a chamber technique in tropical peatland (Jauhiainen *et al.*, 2012), and this was not on

89 burnt peat. Therefore, we continuously measured soil CO_2 efflux using an automated system
90 with six chambers on a burnt peat surface in the former MRP area in Central Kalimantan in
91 2004-2005 under the condition of no plant roots and no leaf litter to directly measure
92 oxidative peat decomposition. Using the continuous data from burnt peat, we assess annual
93 CO_2 emissions arising from peat decomposition and discuss the effects of GWL and peat
94 temperature on peat decomposition.

95

Materials and Methods

Study site

The study was conducted on tropical peatland in the upper catchment of the Sebangau River near Palangkaraya, Central Kalimantan province, Indonesia. In Central Kalimantan, a large peatland area was deforested and drained during the MRP in the late 1990's mainly to develop farmland. Although the project was terminated in 1999, it left a vast, degraded peatland with an enhanced risk of fires (Hoscilo *et al.*, 2011; Page *et al.*, 2009).

The study site (2.34ºS, 114.04ºE), which is in the northern part of Block C of the MRP, was originally covered with peat swamp forest; however, it was burnt in 1997 and 2002, El Niño years, and vegetation and surface peat soil were lost (Hoscilo *et al.*, 2011). The site was called DB in our previous paper (Hirano *et al.*, 2012). The thickness of peat burnt away in 2002 was measured to be 0.22±0.12 m (mean ± one standard deviation (SD) of 25 points) for an area of 25×25 m^2 at this site. Although we have no data on the fire in 1997, Page *et al.* (2002) estimated a burnt depth of 0.51 m as an average over Central Kalimantan peatlands. According to the estimate, the total loss of surface peat was about 0.7 m in thickness as a result of these two fires. A large canal that was excavated in 1996–1997 has functioned effectively to drain the site (Page *et al.*, 2009). In June-August 2005, however, the canal was blocked at several points by small dams to facilitate hydrological restoration of the ex-MRP area (Jauhiainen *et al.*, 2008). Automated soil chambers were installed around a flux tower at a distance of about 200 m from the canal in April 2004, when fern plants were sparsely re-growing. The ground was studded with small pools, but there were no hummocks remaining, which are typical on the floor of peat swamp forest (Jauhiainen *et al.*, 2005). Coarse woody debris (CWD) from stems, branches and rootstocks remained unburnt on the ground. Fern (*Stenochlaena*, *Blechnum* and *Lygodium* spp.) and sedge (*Cyperus*, *Scleria* and *Eleocharis* spp.) plants had grown up to 0.5 m and covered most of the ground in June 2005.

121 The peat depth was about 4 m in 1999 (Tuah *et al*., 2000).

122 Annual values of precipitation and air temperature (the mean ± one SD) for the nine years
123 of 2002–2010 were 2540 ± 596 mm y^{-1} and 26.2 ± 0.3ºC, respectively; these were measured
124 above a drained peat swamp forest near the site (Hirano *et al.*, 2007). Interannual variation in
125 precipitation was large; the maximum (3750 mm y^{-1}) and minimum (1852 mm y^{-1}) were
126 recorded, respectively, in a La Niña year (2010) and an El Niño year (2002). The dry season
127 generally begins in July and lasts through September (Hirano *et al.*, 2012). However, air
128 temperatures showed neither seasonal nor interannual variation. Monthly mean air
129 temperatures ranged within 1ºC of 25.9ºC in July and 26.8ºC in May.

130

131 *Measurement of oxidative peat decomposition*

132 Soil CO_2 efflux was measured continuously using an automated system with six chambers
133 (Hirano *et al.*, 2009; Sundari *et al.*, 2012) from April 2004 to December 2005 with two
134 interruptions due to system malfunction from August to October in 2004 and flooding from
135 December 2004 to May 2005. The chamber was made of an opaque gray PVC cylinder, 40 cm
136 high, with a 25 cm internal diameter and an opaque gray PVC disc as a lid, which was opened
137 and closed automatically using a motor according to a program in a data logger (CR10X;
138 Campbell Scientific Inc., Lorgan, UT, USA). The chambers were directly inserted 13 cm into
139 the ground to prevent horizontal root invasion. The ground-covering area and effective
140 volume of each chamber were, respectively, 0.0491 m^2 and 0.0133 m^3. An air vent with a bore
141 of 1 mm was set in a chamber wall to synchronize the air pressure inside and outside the
142 chamber. Each chamber was closed for 4 minutes one after another. It took 24 minutes to
143 cycle through a round of six chambers. Then the system was stopped for a subsequent 6
144 minutes. Soil CO_2 efflux was calculated half-hourly from the increasing rate of CO_2
145 concentration in the chamber headspace for the last 3 min of each closing. Chamber air was

146 circulated into an infrared gas analyzer (LI820; Licor Inc., Lincoln, NE, USA) with an air
147 pump, which measured CO_2 concentration every 5 s. The rate of CO_2 increase was calculated
148 by linear fitting using the least-squares method. For quality control, CO_2 efflux data were
149 excluded if the correlation coefficient of the fitting was less than 0.424, at which point the
150 linearity was significant at a 1% level. The gas analyzer was calibrated every 3 months using
151 standard gas with two CO_2 concentrations.

152 Six chambers were installed on the flat bare ground, avoiding small ponds, within 3 m of
153 each other in April 2004, 1.5 years after a large fire. There were no living trees with deep root
154 systems within 7-8 m from the chambers. Plant detritus was removed from the soil surface
155 before chamber installation. No plants existed in the chambers throughout the measurement
156 period. Fern and sedge plants re-growing around the chambers were picked off periodically to
157 prevent roots from invading under the deeply-inserted chamber walls. These precautions
158 ensure that soil CO_2 effluxes were equivalent to oxidative peat decomposition rates (peat
159 respiration: RP) without contamination by root respiration.

160 Volumetric soil water content (SWC) was measured for surface peat of 30-cm thickness
161 using a time-domain-reflectometry (TDR) sensor (CSI615; Campbell Scientific Inc.) at three
162 locations on the surface. The TDR output was calibrated using the conventional oven-drying
163 method (Topp & Ferré, 2002). Soil was sampled at depths of 5, 15 and 25 cm, respectively,
164 near each TDR sensor using a core sampler of 50 cm^3 in volume. Soil sampling was
165 conducted three times in different GWL conditions. Soil temperature was measured at 5 cm
166 depth at one point inside and three points outside the chambers, respectively, using
167 thermocouple thermometers. Signals from the sensors were measured every 30 s; their
168 half-hourly means were recorded using a data logger (CR10X; Campbell Scientific Inc.).
169 Groundwater level (GWL) was measured as the distance from the ground surface every 30
170 minutes using a water level logger (DL/N; Sensor Technik Sirnach AG, Switzerland or

DCX-22 VG; Keller AG, Switzerland). These underground sensors were installed within 3 m from the chambers.

Temperature correction and gap filling

Chambers altered soil temperature in comparison with surrounding soil surfaces. Inside temperatures were higher during the daytime and lower during the nighttime than the outside temperatures (Fig. 1). On average, daily maximum and minimum temperatures increased 3.7ºC and decreased 2.8ºC, respectively, in chambers, which doubled the diurnal range. To remove such a temperature effect, soil CO_2 efflux (RP) was modeled using soil temperature and GWL (Eqn. 1).

$$RP = a \cdot \exp(b \cdot T_i) \qquad (1)$$

where a is base RP at 0ºC, b is a temperature factor and T_i is the soil temperature inside a chamber at 5-cm depth; a and b are linearly related to GWL ($a = c + d \cdot GWL$ and $b = e + f \cdot GWL$); both base RP and temperature factors are related to GWL. The validity of these relationships with GWL will be described later. The fitting parameters of c, d, e and f were determined by nonlinear regression, and then RP was corrected using the following equation (Eqn. 2).

$$RP_c = RP_m \cdot \exp[b \cdot (T_o - T_i)] \qquad (2)$$

where RP_c is temperature-corrected RP, RP_m is measured RP and T_o is outside soil temperature. Eqn. 1 was also used to fill data gaps after quality control.

Results

Effect of groundwater level

To remove the effect of diurnal temperature variation (Fig. 1), daily mean soil CO_2 efflux (RP) was calculated for each chamber if the number of half-hourly data were more than 36 (three quarters) on a day after the quality control. As a result, 156-299 days of daily mean data were generated for each chamber. The RP increased as GWL decreased (Fig. 2). The negative relationship can be approximated significantly ($p < 0.001$) with a logarithmic curve ($y = 1.48 \cdot \ln(1.48 - 5.96 \cdot x)$, $r^2 = 0.64$) or a line ($y = 0.79 - 2.73 \cdot x$, $r^2 = 0.60$); where x and y are GWL (m) and RP (μmol m^{-2} s^{-1}), respectively. The linearity suggests that every GWL lowering by 0.1 m causes an additional RP of 0.27 μmol m^{-2} s^{-1}. However, variation in RP increased as GWL lowered. The deviation from the curve or line could not be explained significantly by daily mean SWC nor soil temperature. The deviation was due mostly to spatial variation in RP.

Effect of soil temperature

Half-hourly data were binned into classes at 0.1-m intervals of GWL to remove the GWL effect (Fig. 2), and then Eqn. 1 was fitted to data in each class using inside soil temperature at a depth of 5 cm on the restrictive assumption that coefficients a and b are independent of GWL within its small range. On the whole, the temperature factor (b) shows a significant positive relationship with GWL ($p < 0.01$, Fig. 3). According to the linear relationship, Q_{10} was estimated to be 1.36 and 1.16 at GWLs of 0 and -0.6 m, respectively. Figures 2 and 3 assure the linear relationship of the coefficient of a or b with GWL in Eqn. 1, respectively.

Table 1 lists fitting parameters in Eqn. 1 for each chamber. All parameters are statistically significant ($p < 0.001$). Although parameters differ among chambers, GWL and surface soil temperature accounted for 47 to 61% (r^2 = 0.47 to 0.61) of total variation in RP on a

half-hourly basis. Soil temperature or GWL was solely used as variables to fit a simple exponential equation or a linear equation to half-hourly data, respectively. The r^2 values of the exponential curve and the line were 0.001-0.05 and 0.38-0.55, respectively. The result shows half-hourly RP depended much more on GWL than soil temperature and the combination of the two variables improved model performance. Eqn. 2 was applied to make a temperature correction of measured RP on a half-hourly basis. The mean values (± one SD of six chambers) of measured and corrected RP (n = 12745 to 15053) were 1.35±0.24 and 1.37±0.25 μmol m^{-2} s^{-1}, respectively, for the whole measurement period. The small difference of 0.02 μmol m^{-2} s^{-1} or 1.5% suggests that temperature disturbance due to chamber installation hardly affected the mean and total RP values.

Seasonal variation

Seasonal variations in daily means of soil temperature, GWL, SWC and temperature-corrected RP are shown in Fig. 4. The RP varied seasonally in reverse parallel with GWL variation. The RP clearly increased during the dry season with low GWL. The SWC of the surface layer with a thickness of 30 cm synchronized with GWL, whereas SWC was saturated when GWL rose aboveground. Soil temperature showed a positive linear relationship with GWL ($p < 0.001$) (data not shown). Figure 5 also shows seasonal variation in temperature-corrected RP after gap filling together with ecosystem respiration (RE) measured by the eddy covariance technique at the same site (Hirano *et al.*, 2012). The seasonal variation of RP was much larger than that of RE (Fig. 5b). The RP contributed more to RE in the dry season (Fig. 5c). The contribution of RP increased up to 40% or more in the dry season, whereas it was less than 20% in the rainy season.

Annual soil CO_2 efflux

The RP in Fig. 5 was summed for two annual periods from DOY (day of year) 108 in 2004 to DOY 107 in 2005 (Year 04-05) and DOY 108 in 2005 to DOY 107 in 2006 (Year 05-06), respectively (Table 2). Although RP measurement was finished in December 2005, RP was calculated half-hourly from GWL and soil temperature using Eqn.1 for 107 days in 2006. In the 04-05 and 05-06 years, 58-70% and 56-75% of total half-hourly data were missed, respectively, and the data gaps were filled for each chamber using Eqn. 1 (Table 1) from soil temperature and GWL. The percentage of data gaps depended on chambers. Annual RP was also calculated half-hourly using Eqn. 1 for every annual period from DOY 108 in 2004 to DOY 107 in 2009 (Table 2). Measured (+ gap filled) and modelled annual RP was 382±82 (the mean ± one SD of six chambers) and 393±104 gC m^{-2} y^{-1} in year 04-05, and 362±74 and 335±60 gC m^{-2} y^{-1} in year 05-06, respectively. On average, modelled RP was 11 gC m^{-2} y^{-1} (2.9%) larger than measured RP in year 04-05, whereas it was 27 gC m^{-2} y^{-1} (7.5%) smaller in year 05-06. Measured RP accounted for 24 and 21% of RE in 04-05 and 05-06 years, respectively. Modelled RP shows a significant negative relationship with GWL ($p < 0.01$) on an annual basis, which suggests that RP increases by 89 gC m^{-2} y^{-1} every time annual mean GWL lowers by 0.1 m. This rate on an annual basis can be converted to 0.24 μnol m^{-2} s^{-1} and is compatible with the measured rate of 0.27 μnol m^{-2} s^{-1} (Fig. 2).

Residual of ecosystem respiration after the subtraction of peat decomposition

The residual of the subtraction of RP from RE (RE – RP) corresponds to the combination of autotrophic respiration and decomposition of CWD and litter. The residual varied seasonally and was small in the dry season and large in the rainy season (Fig. 5). A positive linear relationship was found ($p < 0.001$) between the residual and GWL when GWL was belowground (Fig. 6). However, the residual decreased as GWL increased ($p < 0.001$) when GWL rose aboveground. The annual sum of the residual was 13% larger in year 05-06 than

year 04-05 (Table 2).

Discussion

Effect of peat burning

This study was conducted on a burnt peat area, in which surface peat was likely lost by about 0.7 m in total thickness through large-scale fires. Consequently, CO_2 efflux was measured on well-decomposed deeper peat with more recalcitrant carbon (Bozkurt *et al.*, 2001; Limpens *et al.*, 2008; Page *et al.*, 2004), which underlay the surface peat lost by the fires. The loss of surface peat, which was fresher and less decomposed, probably reduced oxidative peat decomposition (RP) in comparison with that on unburnt peat. In addition, fires alter peat properties chemically, physically and microbiologically. According to field studies in mid- and high latitudes, as a result of combustion, organic carbon and total nitrogen decrease in the remaining peat, whereas mineral nutrients, pH and bulk density increase (Dikici & Yilmaz, 2006; Smith *et al.*, 2001). The increase in pH can enhance soil microbial activity and potentially enhances peat decomposition (Moilanen *et al.*, 2012). However, although the application of wood or peat ash increased peat pH, its effect on RP is complex; there are contradictive studies reporting positive (Moilanen *et al.*, 2012), negative (Klemedtsson *et al.*, 2010) and variable (Hogg *et al.*, 1992) effects. Further investigation is necessary with enough field data to investigate the effect of peat burning on RP, because burning effect on the belowground environment is variable, depending on burnt frequency and fire severity (Neary *et al.*, 1999).

Effect of soil temperature and chamber installation

The effect of soil temperature was analyzed using temperature data measured at a depth of 5 cm. The apparent temperature sensitivity of RP tended to decrease as GWL lowered (Fig. 3), mostly because the vertical distribution of peat decomposition changed along with GWL

change (Laiho, 2006). Apparent Q_{10} was calculated at 1.36 and 1.16 at GWLs of 0.0 and -0.6 m, respectively, from the linear relationship (Fig. 3). The contribution of the decomposition of deeper peat to soil CO_2 efflux (RP) would increase as GWL lowers, because deeper peat is more aerated with lower GWL. Thus, low GWL shifted the vertical distribution of peat decomposition downward and deepened the peak position of peat decomposition, which separates the peak depth of peat decomposition from the temperature-measuring depth. As a result, RP became more out of phase with soil temperature at 5-cm depth as GWL lowered. Temperature sensitivity probably increases, if soil temperature was measured at a peak depth. The uppermost layer of several centimeters in thickness dried out in the late dry season, whereas mean SWC of the upper layer of 30-cm thickness kept relatively high above 0.52 m^3 m^{-3} (Fig. 4). The surface desiccation probably reduced the apparent temperature sensitivity (Davidson & Janssens, 2006; Suseela *et al.*, 2012). Although high GWL showed a larger temperature factor, the absolute effect of temperature was very limited because of small RP at high GWL (Fig. 2). A similar relationship of temperature sensitivity with GWL was reported for heterotrophic peat soil respiration in drained peatlands in Finland (Mäkiranta *et al.*, 2009), although in that study the data were obtained from several sites. Hirano *et al.* (2009) showed apparent Q_{10} values above 2.4 for soil temperature at a depth of 5 cm, for total soil respiration at drained and almost undrained tropical peat swamp forests near this site. However, the Q_{10} would be strongly affected by a circadian rhythm in root respiration (Hirano *et al.*, 2009) and much less for heterotrophic respiration. In addition, root respiration has a higher Q_{10} than heterotrophic respiration (Boone *et al.*, 1998).

The study site used to be a swamp forest but was cleared of trees by fires. As a result, soil temperature at a depth of 5 cm was 2.9ºC higher on a daily basis and had larger diurnal range by 3.4ºC than that in an almost undrained swamp forest (Sundari *et al.*, 2012) on average for 515 days in 2004 and 2005; the forest site was about 15 km apart from the burnt site. The

GWLs were very similar for each site (Hirano *et al.*, 2012). Using models determined for each chamber (Eqn. 1), RP was estimated from half-hourly data of soil temperature in the forest site and GWL in the study site to investigate the effect of temperature change following deforestation on RP. The result shows that the means of measured and estimated RP for the 515 days were 1.24±0.27 and 1.19±0.25 μmol m^{-2} s^{-1}, respectively. The higher temperature increased RP by 4% on average. The comparison suggests that the effect on peat decomposition of temperature rise by forest clearance is small in burnt sites.

Chamber installation increased the diurnal range of soil temperature from 5.9 to 12.5ºC on average, whereas daily means were similar between inside and outside chambers: 28.8 and 28.3ºC, respectively (Fig. 1). In addition, such temperature disturbance attenuates with peat depth. These facts suggest that the chamber effect through temperature alteration (Fig. 1) was very limited on the whole. Chambers may have reduced evaporation from the soil surface, which increases moisture of shallow soil. However, we have no data to assess moisture alteration by chambers.

Effect of groundwater level

Daily-mean RP increased as GWL decreased, and the relationship was approximated significantly ($p < 0.001$) with a logarithmic curve ($r^2 = 0.64$) or a line ($r^2 = 0.60$) (Fig. 2). The line suggests that daily RP increases by 0.27 gC m^{-2} d^{-1} every GWL lowering of 0.1 m. The shape of the relationship resembles that of total soil respiration in the almost undrained forest (Sundari *et al.*, 2012). The enhancement of RP or oxidative peat decomposition under the low GWL conditions was due to thickening of an unsaturated soil zone and its resultant enhancement of aeration. In contrast, no such relationship was found for instantaneous oxidative peat decomposition in an Acacia plantation in Sumatra, Indonesia (Jauhiainen *et al.*, 2012), in which GWL was controlled. The obscure relationship with GWL was attributed to

342 drainage. The Acacia plantation was drained and GWL kept underground. Although our site
343 was drained, GWL was almost the same as that of an undarined forest (Hirano *et al.*, 2012)
344 because the fire-induced subsidence brings the peat surface closer to the water table.

345 Annual oxidative decomposition of tropical peat was linearly related to GWL using
346 chamber data (Jauhiainen *et al.*, 2012) in an Acacia plantation and subsidence data (Hooijer *et*
347 *al.*, 2012) in the same Acacia plantation and an adjacent drained natural forest. The
348 relationships show that every 0.1 m lowering of annually mean GWL enhances peat
349 decomposition by 195 and 188 gC m^{-2} y^{-1} for an Acacia plantation from chamber and
350 subsidence data, respectively, and 267 gC m^{-2} y^{-1} for a drained forest from subsidence data. In
351 addition, a result of meta-analysis of subsidence data suggests that peat decomposition
352 increased by at least 250 gC m^{-2} y^{-1} for each 0.1 m of additional GWL lowering (Couwenberg
353 *et al.*, 2009). In contrast, our model simulation suggests that RP increases by 89 gC m^{-2} y^{-1}
354 every time annual mean GWL lowers 0.1 m (Table 2). On an annual basis, peat oxidative
355 decomposition is less sensitive to GWL in the burnt site than in the other unburnt tropical
356 peatlands. The smaller GWL sensitivity is attributable to the loss of surface peat, which is
357 more decomposable (Hogg *et al.*, 1992).
358
359 *Contribution of peat decomposition to ecosystem respiration*
360 The contribution of RP to ecosystem respiration (RE) had an obvious seasonality (Fig. 5),
361 which shows that the contribution increased in dry conditions. On an annual basis, RP
362 accounted for 24% and 21% of RE in 04-05 and 05-06 years, respectively (Table 2). In
363 contrast, the residual of the subtraction of RP from RE, which corresponds to the combination
364 of autotrophic respiration and decomposition of CWD and litter, decreased in dry conditions
365 and increased in moist conditions, whereas it decreased if GWL rose aboveground (Fig. 6).
366 The GPP or ecosystem photosynthesis showed a quadratic relationship with GWL with low

GPP both at low and high GWLs in this site (Hirano *et al.*, 2012). Autotrophic respiration would respond to GWL similarly, because autotrophic respiration is strongly linked to GPP through direct consumption of photosynthate. The positive or optimum moisture responses of the decomposition rates of CWD (Jomura *et al.*, 2007; Wang *et al.*, 2002) and litter (Kim *et al.*, 2005) also support the relationship between the residual and GWL (Fig. 6).

Comparison in annual CO_2 efflux with other sites

Annual RP was assessed at 382 and 362 gC m^{-2} y^{-1} in 04-05 and 05-06 years, respectively (Table 2), which were about 27-28% of annual soil respiration (1347 gC m^{-2} y^{-1}), including root respiration, measured in 2005 in an almost undrained peat swamp forest with similar GWL (Sundari *et al.*, 2012). In addition, the annual RP was only 17% of peat decomposition measured in an Acacia plantation in Sumatra (2182 gC m^{-2} y^{-1}) with annually mean GWL of -0.8 m (Jauhiainen *et al.*, 2012). The large difference in peat decomposition between the two sites is chiefly attributable to GWL difference. Linear relationships between peat decomposition and GWL on an annual basis were determined for an Acacia plantation and a neighboring drained natural forest from chamber data (Jauhiainen *et al.*, 2012) and subsidence and bulk density data (Hooijer *et al.*, 2012). Using the relationships, annual RP was simply estimated for annually mean GWLs of -0.18 and -0.09 m, respectively, which corresponded to those at the burnt site in 04-05 and 05-06 years, respectively (Table 2). Results show that peat decomposition in the Acacia plantation was 911-982 and 742-807 gC m^{-2} y^{-1} for GWLs of -0.18 and -0.09 m, respectively. The annual RP in the Acacia plantation was two to three times as large as that in the burnt site even at the same GWL. However, the estimates may have considerable uncertainties because of extrapolation beyond the range of measured GWL. On the other hand, annual RP in the drained natural forest was 481 and 241 gC m^{-2} y^{-1} for GWLs of -0.18 and -0.09 m, respectively. The annual estimates were compatible with those in the

burnt site. This compatibility suggests that the effect of peat burning on oxidative peat decomposition is not large at relatively high GWL close to the ground surface.

Implications for carbon emissions from fire-degraded tropical peatlands

There is little information on CO_2 emissions through oxidative peat decomposition from fire-degraded tropical peatlands. Although the decomposition of degraded peat is expected to be chiefly controlled by GWL and temperature, it is difficult to simply assess the CO_2 emissions using such physical factors on a large-scale, because peat degradation depends on fire conditions, such as severity and frequency. Here, we try to generalize the fire effects on the decomposition of severely-degraded peat from a viewpoint of GWL and temperature.

During fires, upper peat with the higher proportion of labile carbon burns preferentially. The loss of upper peat potentially reduces oxidative CO_2 emissions, because remaining peat, which undelay the upper peat until fires, has more recalcitrant compounds. In addition, the ground subsides sharply owing to peat loss, and consequently GWL rises relatively. Thus, the subsidence should reduce peat decomposition with higher GWL. Fires clear vegetation from peat swamp forest and open the ground in many cases. As a result, soil temperature rises especially in the surface soil layer, and peat decomposition theoretically accelerates. In an open area, however, surface peat can easily desiccate in the dry season. Also, peat decomposition tended to be decoupled with surface soil temperature as GWL lowers. Although peat decomposition was coupled with soil temperature at high GWL, the level of CO_2 emissions was low in such water-saturated conditions. Thus, the effect of temperature rise by land opening on the decomposition of fire-degraded peat is probably limited.

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535

536 **Table1** Fitted parameters of Eqn. 1.

	c	*d*	*e*	*f*	r^2	*n**
Chamber 1	0.327	-3.32	0.0283	0.0159	0.58	14240
Chamber 2	0.190	-3.05	0.0397	0.0218	0.55	14604
Chamber 3	0.172	-1.73	0.0375	0.0318	0.61	14287
Chamber 4	0.484	-4.31	0.0199	0.0387	0.55	15053
Chamber 5	0.264	-1.60	0.0344	0.0192	0.57	12958
Chamber 6	0.352	-5.16	0.0236	0.0500	0.47	12745
Mean	0.298	-3.19	0.0306	0.0296	0.56	13981
SD*	0.116	1.40	0.0079	0.0131	0.05	924

537

538 *Standard deviation

539 **Number of data

540

Table 2 Annual soil CO_2 efflux (RP) through peat decomposition and ecosystem respiration (RE) (gC m^{-2} y^{-1})

Annual period*	GWL (m)	RP						Mean	SD***	RE****	RE - RP	RP / RE
		Chamber 1	Chamber 2	Chamber 3	Chamber 4	Chamber 5	Chamber 6					
04-05	-0.18	480 (505)**	467 (516)	295 (285)	411 (419)	329 (285)	309 (348)	382 (393)	82 (104)	1617	1235 (1224)	0.24 (0.24)
05-06	-0.09	412 (388)	422 (383)	260 (245)	391 (378)	274 (282)	412 (336)	362 (335)	74 (60)	1762	1400 (1427)	0.21 (0.19)
06-07	-0.24	(556)	(560)	(298)	(430)	(275)	(330)	(408)	(128)	1849	(1441)	(0.22)
07-08	-0.01	(239)	(208)	(160)	(261)	(183)	(201)	(209)	(37)	1831	(1622)	(0.11)
08-09	-0.01	(245)	(237)	(156)	(252)	(183)	(221)	(216)	(38)	1795	(1579)	(0.12)
Mean	-0.11	(386)	(379)	(234)	(349)	(249)	(294)	(315)	(72)	1771	(1459)	(0.18)
SD***	0.10	(142)	(151)	(71)	(86)	(64)	(86)	(96)	(37)	92	(156)	(0.06)

*From DOY108 to DOY107 in the following year.

**Figures in parentheses are calculated using the model (Eqn. 1) from soil temperature and groundwater level (GWL).

***Standard deviation.

****Measured by the eddy covariance technique (Hirano et al., 2012).

Fig. 1. Diurnal variations in soil temperature inside and outside chambers. Half-hourly data were ensemble-averaged for the whole measurement period. Vertical bars denote 1 standard deviation.

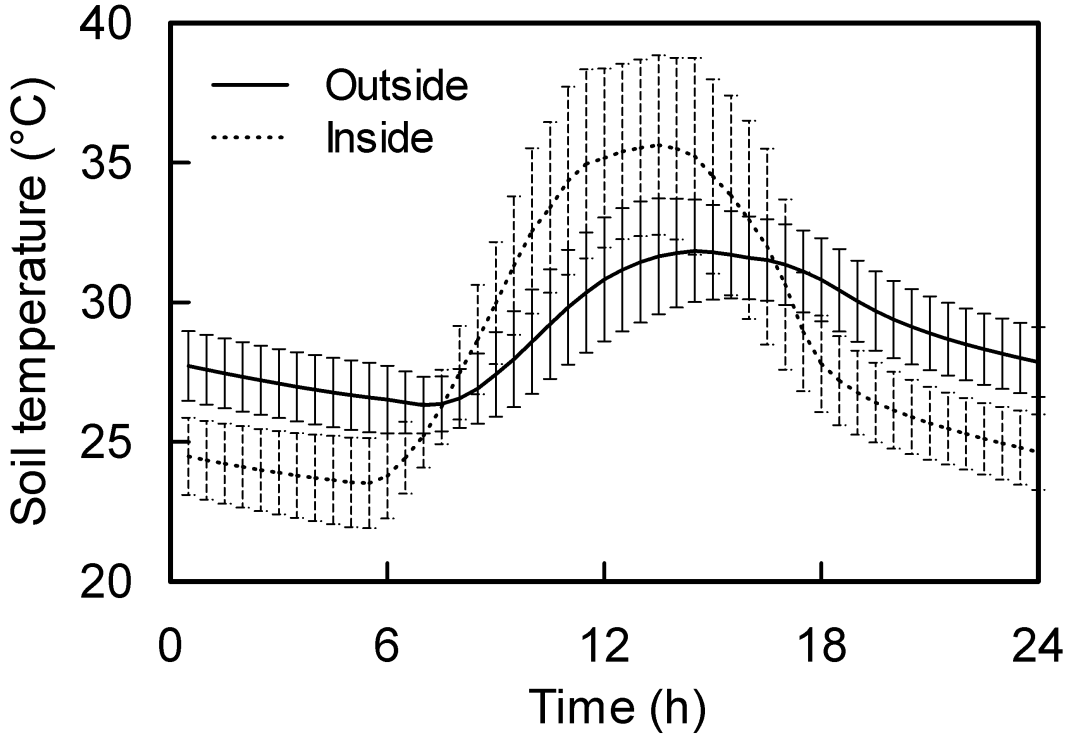
Fig. 2. Relationship between daily means of soil CO_2 efflux (RP) through peat decomposition and groundwater level (GWL). A fitted logarithmic curve (r^2 = 0.64) and a fitted line (r^2 = 0.60) are drawn in solid and broken, respectively.

Fig. 3. Relationship between temperature factor (*b*) in Eqn. 1 and groundwater level (GWL). The temperature factor of each chamber was determined by fitting Eqn. 1 to data binned into classes at 0.1-m intervals of GWL, on the assumption that coefficients *a* and *b* are independent of GWL. Different symbols denote different chambers. A positive line was fitted (r^2 = 0.15).

Fig. 4. Seasonal variations in daily means of soil temperature outside chambers (a), groundwater level (GWL) (b), soil moisture (SWC) (c) and soil CO_2 efflux (RP) through peat decomposition (d) in 2004 and 2005. Different symbols denote different chambers. Horizontal bars in Fig. 4d denote the dry season determined by monthly precipitation.

Fig. 5. Seasonal variations in daily soil CO_2 efflux (RP) through peat decomposition after gap filling (a), mean RP and ecosystem respiration (RE) (b) and the ratio of RP and RE (c) in 2004 and 2005. Different symbols denote different chambers.

Fig. 6. Relationship between the residual of the subtraction of soil CO_2 efflux (RP) through peat decomposition from ecosystem respiration (RE) (RP – RE) and groundwater level (GWL) on a daily basis. Different lines were fitted to data sets at GWLs lower (r^2 = 0.73) and higher (r^2 = 0.22) than the ground level, respectively.



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576 Fig. 1

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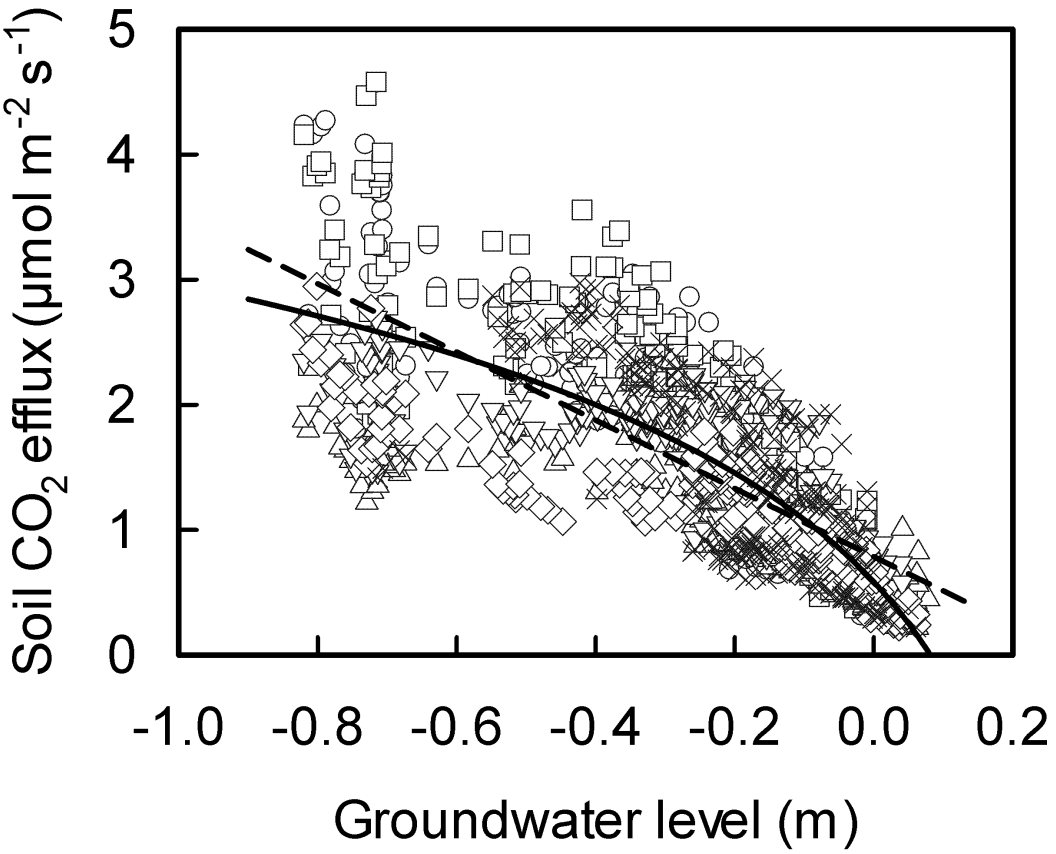

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579 Fig. 2

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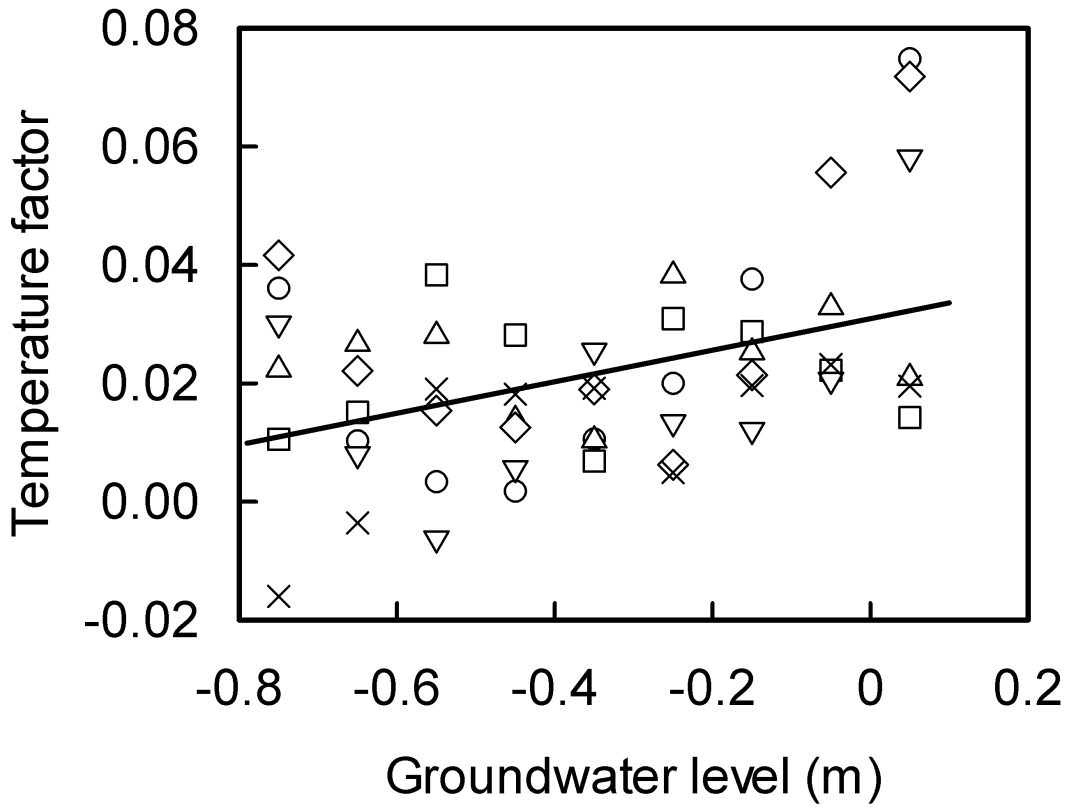

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583 Fig. 3

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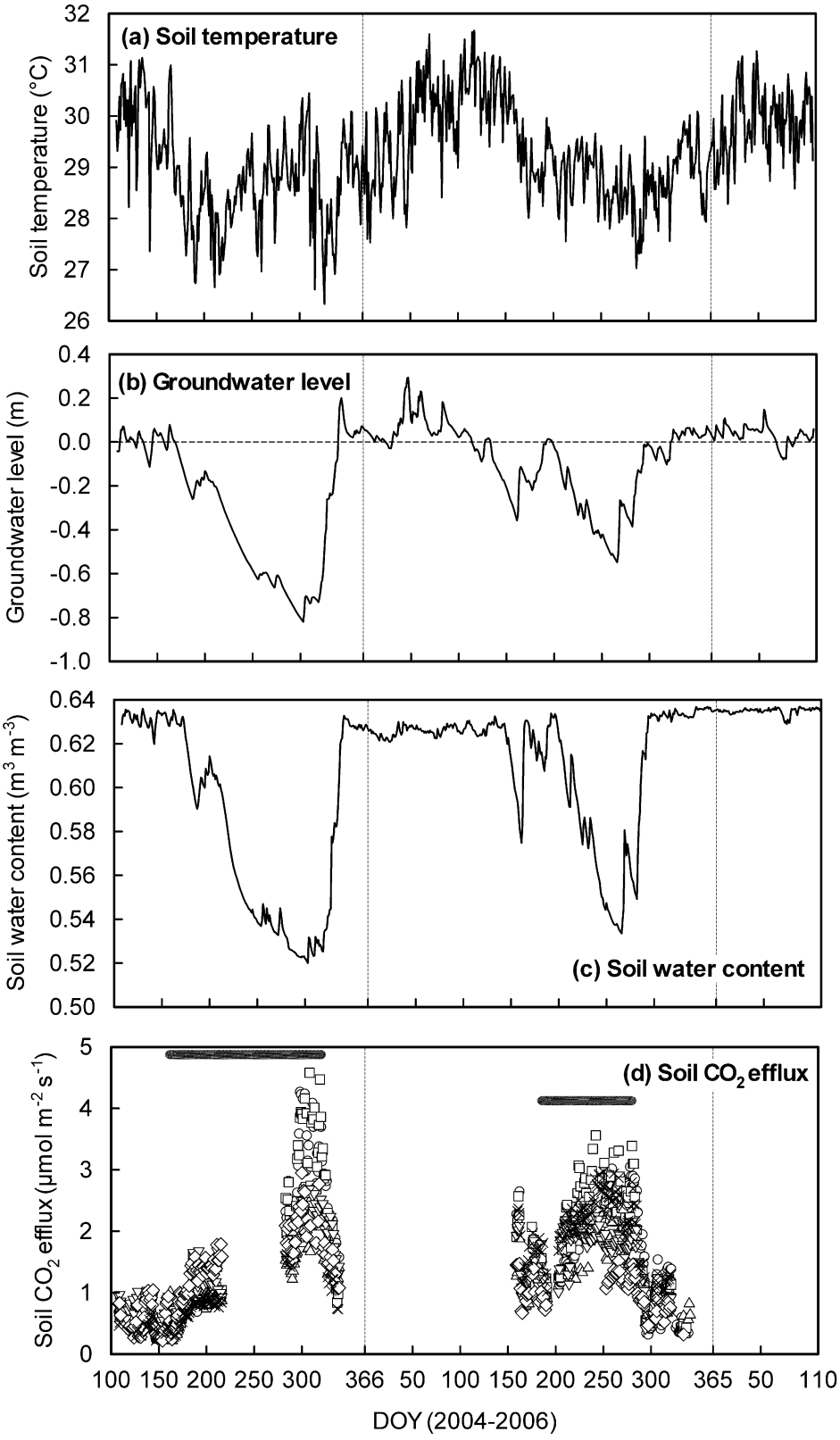

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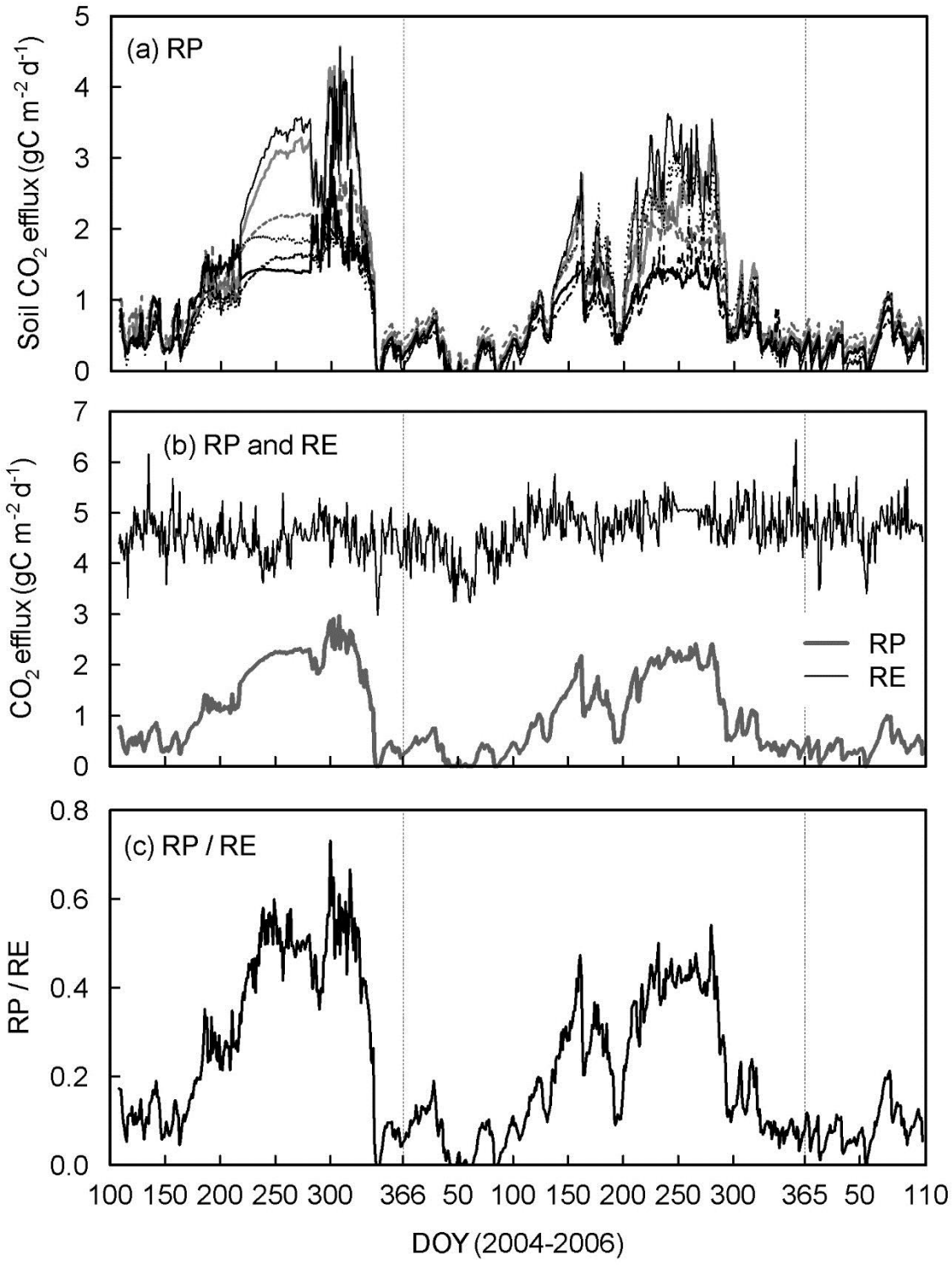
587 Fig. 4



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590 Fig. 5

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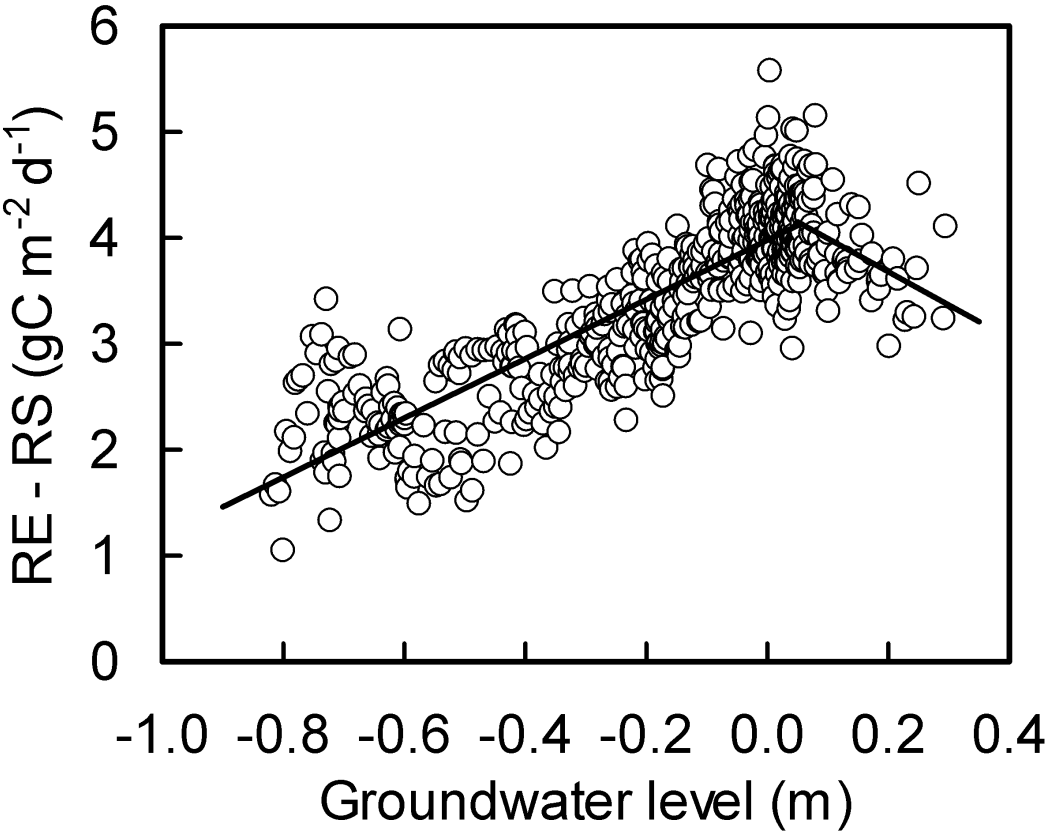

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594 Fig. 6