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Effect of repeated fires on land-cover change on peatland in southern Central Kalimantan, Indonesia, from 1973 to 2005

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Abstract. Fire plays an increasingly important role in deforestation and degradation of carbon-dense tropical peatlands in South-east Asia. In this study, analysis of a time-series of satellite images for the period 1973–2005 showed that repeated, extensive fires, following drainage and selective logging, played an important role in land-cover dynamics and forest loss in the peatlands of Central Kalimantan, Indonesia. A study of peatlands in the former Mega Rice Project area revealed a rising trend in the rate of deforestation and identified fire as the principal factor influencing subsequent vegetation succession. A step change in fire regime was identified, with an increase in burned area and fire frequency following peatland drainage. During the 23-year pre-Mega Rice Project period (1973–1996), peat swamp forest was the most extensive land-cover class and fires were of relatively limited extent, with very few repeated fires. During the 9-year post-Mega Rice Project period (1997–2005), there was a 72% fire-related loss in area of peat swamp forest, with most converted to non-woody vegetation, dominated by ferns or mosaics of trees and non-woody vegetation, rather than cultivated land.

Additional keywords: burned area detection, carbon loss, deforestation, Mega Rice Project, peat swamp forest, remote sensing, tropical peatland.

Introduction

Tropical forests cover only 7–10% of the Earth's surface but they are globally important, containing 40–50% of all carbon stored in terrestrial vegetation (Malhi and Grace 2000; Nightingale *et al*. 2004). The role of these forests in the global carbon cycle is important because it is estimated that tropical deforestation is responsible for ~20% of global anthropogenic carbon emissions (Fischlin *et al.* 2007). A recent study by Hansen *et al*. (2009) estimated that the Sumatran and Kalimantan lowlands experienced an increase in the annual rate of deforestation from 1.3% year^{-1} in 1990–2000 to 1.4% year^{-1} in 2000–05. Some of the highest losses in the lowland areas were associated with the peat swamp forest ecosystem. Hooijer *et al*. (2006), for example, showed that between 2000 and 2005, these wetland forests were being lost at rates of 1.5 and 2.2% year^{-1} in Malaysia and Indonesia respectively. The principal drivers of rapid peatland deforestation are agricultural expansion, particularly of oil-palm and pulpwood plantations, and timber extraction (Geist and Lambin 2002; Butler *et al.* 2009; Danielsen *et al.* 2009), but forest disturbance has also increased the risk of fire, leading to further loss and fragmentation of the region's remaining forests (Siegert *et al.* 2001; Dennis *et al.* 2005). Fires, however, are not a totally new phenomenon in tropical forests and neither are they entirely dependent on human activity. Lightning strikes have been noted as a natural source of ignition for tropical forest fires, although these events are usually associated with heavy rainfall and thus rarely lead to forest fires (Tutin *et al.* 1996). Langner *et al.* (2007) showed that fire plays an important role in deforestation on the island of Borneo, and is strongly correlated with land-cover change. Peat swamp forest is particularly vulnerable to fire; in 2002, for example, 73% of the forest area of Borneo affected by fire was peat swamp forest whereas in 2005, it was 55% (Langner *et al.* 2007). Over the 10-year period 1997–2006, the peat-covered lowlands of Central Kalimantan were one of the most fire-affected landscapes in Borneo, with evidence of multiple fires (Langner and Siegert 2009).

The largest extent of tropical peat-covered lowland, with ~56% of the total global area, occurs in South-east Asia, and almost half of this area (47%, equivalent to 206 950 km^2) is located in Indonesia (Page *et al.* 2011). Tropical peatland is a carbon-dense ecosystem containing globally a peat carbon pool of 88.6 Gt (equal to 15–19% of the global peat carbon pool), of which 57.4 Gt carbon is in Indonesian peatlands alone (Page *et al.* 2011). Carbon accumulation in an undisturbed peat swamp forest ecosystem is maintained by a strong interrelation between forest vegetation and the peat under anoxic, waterlogged conditions (Page *et al.* 2006). Most anthropogenic activities, especially timber extraction, forest clearance, peatland drainage and fires (both natural and caused by humans) lead to carbon losses from the forest and peat carbon pools and significant emissions of CO_2 to the atmosphere (Page *et al.* 2002; Hooijer *et al.* 2006; Jauhiainen *et al.* 2008; Rieley *et al.* 2008; Joosten

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and Couwenberg 2009). The devastating 1997–98 Indonesian peatland fire, for example, was one of the largest peak emission events in the recorded history of fires in equatorial South-east Asia and globally (van der Werf *et al*. 2006; Schultz *et al*. 2008). Page *et al*. (2002) estimated conservatively that more than 0.87 Gt of carbon were lost in this one fire event, equivalent to 14% of the average annual global fossil fuel emissions released during the 1990s. This scale of emission was confirmed by Schultz *et al*. (2008) and van der Werf *et al*. (2008), who respectively estimated 1997 emissions at 1.14 Gt of carbon from Indonesia and 1.09 Gt of carbon from equatorial South-east Asia, over 90% of which originated from Indonesia. This 'mega-fire event' led to an increased awareness of the wide-ranging effects that uncontrolled fires in this region, and in particular peatland fires, have on biodiversity, economy, human wellbeing and climate (Aiken 2004; Curran *et al*. 2004; Slik *et al*. 2008).

The Mega Rice Project (MRP) is an example of anthropogenic pressure on tropical peatland areas expressed through intensive peat drainage. The MRP was established in Central Kalimantan by the Government of Indonesia in 1995 to convert $\sim 1 \times 10^6$ ha of tropical forest, mostly forested peatland (61% of the MRP area) into cultivated land, particularly rice fields. The project started in 1996 with a phase of intensive infrastructure development, involving construction of a system of more than 4500 km of drainage and irrigation channels within each of five working Blocks A–E (Notohadiprawiro 1998; Muhamad and Rieley 2002). After a few years, it became clear that the project was failing owing to the acidic, infertile peat, the malfunctioning water-control system and the difficulties faced by transmigrant farmers attempting to undertake any form of economic agriculture (Muhamad and Rieley 2002; Sulistiyanto 2004). The project was officially terminated in 1999 and most of the land was subsequently abandoned, leaving a degraded landscape, with a network of malfunctioning drainage canals and over-drained, rapidly oxidising peat soils that are susceptible to fire. Previous studies have highlighted that the MRP area is now very fire-prone (Boehm and Siegert 2001; Page *et al*. 2002; Langner and Siegert 2009), with implications for loss of the peat carbon store, biodiversity and other valuable natural resource functions (Page *et al*. 1999). In their studies, Langner *et al*. (2007) and Langner and Siegert (2009) adopted a regional approach to investigating the role of fire in forest loss and degradation over the period 1997–2006 using low- and medium-resolution optical satellite data from the Advanced Very High Resolution Radiometer (AVHRR) on the National Oceanic and Atmospheric Administration (NOAA) satellite and Moderate Resolution Imaging Spectroradiometer (MODIS) carried on board of the Terra and Aqua satellites (spatial resolution 1 km to 250 m). These sensors are unable to detect small areas of land cover and forest change that can be observed from high-resolution satellite sensors, e.g. Landsat or Disaster Monitoring Constellation (spatial resolution ~30 m) (Hansen *et al*. 2008). In addition, Landsat imagery provides the longest and the most comprehensive observation of land-cover dynamics associated with tropical deforestation at regional and global scales.

We used a time-series of high-resolution Landsat and Disaster Monitoring Constellation data to provide a detailed, long-term (32-year period) assessment of the peat swamp forest ecosystem in the former MRP area, in order to investigate (i) land-cover change and (ii) fire regime, and (iii) to understand the effect on the environment of the anthropogenic activities associated with the implementation of the MRP.

Methods

Study area

The study was carried out in Block C, the western-most part of the former MRP area, which covers ~4500 km^2 (Fig. 1) and comprises almost 40% of the total peatland area within the MRP (Rieley and Page 2005). Block C contains an entire peat dome bounded by the Kahayan and the Sebangau Rivers to the east and west respectively, the Java Sea to the south and Palangka Raya, the provincial capital of Central Kalimantan, to the north. This region has a wet tropical climate with annual rainfall between 2000 and 3000 mm that exceeds the rate of evaporation except during the driest months (from June to September) and El Niño events.

Lowland tropical peatlands have a characteristically dome-shaped surface, which is related to their ombrotrophic (rain-fed) peat formation (Page *et al*. 2006) that provides a substrate of low nutrient concentration and high acidity (Sulistiyanto 2004). These chemical and physical conditions influence the structure of the peat swamp forest (PSF) vegetation and its biodiversity (Page *et al*. 1999). In natural, undisturbed PSF, at the margins of the dome, species-diverse, closed canopy, mixed peat swamp forest (mixed-PSF, up to 45 m tall) occurs on peat less than 3 m thick, containing several commercially valuable trees such as *Shorea* spp. and *Gonystylus* spp. Further from the rivers, towards the centre of the dome where the peat surface is almost flat, peat is thicker (6 to 10 m thick) and tree canopy height much lower (up to 25 m). This low pole peat swamp forest (low-PSF) is less diverse, with few trees of commercial size (Page *et al*. 1999; Rieley and Page 2005). On alluvial soils along river valleys, PSF grades into freshwater swamp forest, while marine fringes support mangrove forest. In the southern part of the study area, PSF is replaced by heath forest, which is typical of heavily leached sandy substrates.

Land-cover mapping

Land-cover maps were derived from a time-series of satellite images obtained from several Landsat sensors, including Multispectral Scanner (MSS), Thematic Mapper (TM) and Enhanced Thematic Mapper (ETM+). A series of Landsat images was used to map the land cover over a period of 32 years from 1973 until 2005. An orthorectified Landsat ETM+ image acquired in 2000 was used as a reference to co-register other non-projected images. All images were projected to the Universal Transverse Mercator (UTM) projection, Zone 50 South, with a total residual root-mean-square error of less than a half pixel. Owing to the heterogeneous character of the landscape and variation in the spatial and spectral resolution of the available data, differences in acquisition dates and data quality, a visual, expert-based interpretation supported by ecological knowledge, obtained during the field reconnaissance mission, was chosen as the method that could provide the most consistent and comparable results. The validation of the land-cover maps was only possible for the most recent sampling year (2005).

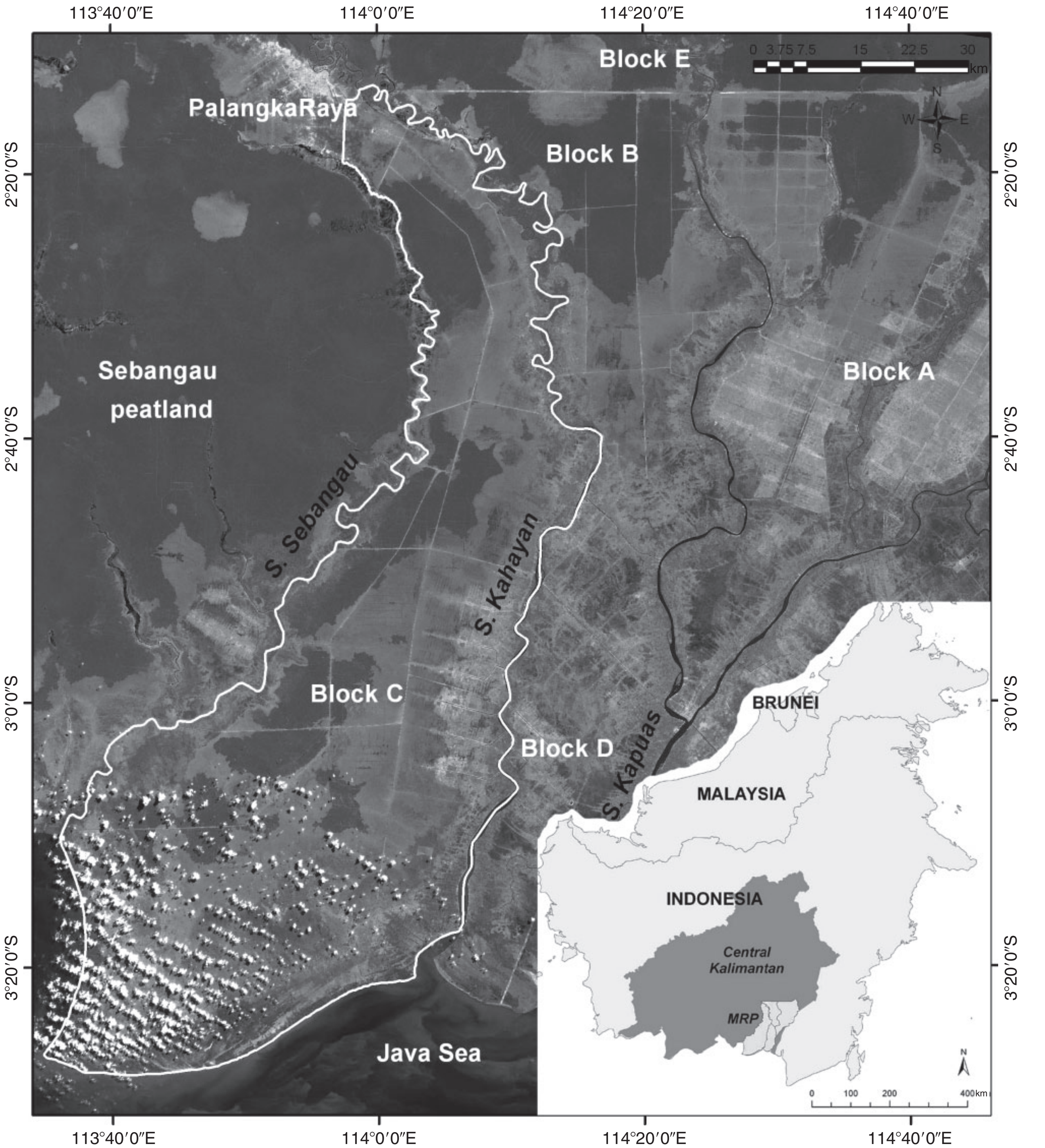

Fig. 1. Study area: Block C is located in the western part of the ex-Mega Rice Project (MRP), in Central Kalimantan, Indonesia.

It was performed using ground-truth data collected during field missions in 2005 and 2006.

The land-cover legend was based on the widely used Tropical Ecosystem Environment Observation by Satellite (TREES) classification scheme, which was modified specifically for the insular South-east Asia region by Stibig *et al*. (2003, 2007). The TREES land-cover categories were amended for the present study to provide an appropriate subregional context as well as more specific detailed delineation of vegetation classes that might be important for analysis of post-fire vegetation regrowth.

Table 1. Categories of land-cover classes used in this study of Block C of the former Mega Rice Project area in Central Kalimantan, Indonesia
TREES, Tropical Ecosystem Environment Observation by Satellite

TREES classification	Land-cover classification used in this study	
	Classes	Subclasses
Forest cover		
Swamp forest	Peat swamp forest (PSF)	Mixed-PSF
		Low pole-PSF
		Heavily logged PSF
		Secondary forest
	Freshwater swamp forest	
Mangrove forest	Mangrove forest	Mangrove forest
		Fragmented and degraded mangrove forest
Heath forest	Heath forest	
Mosaic of tree cover and other natural vegetation	Mosaic of trees and non-woody vegetation	
Non-forest vegetation cover		
Evergreen shrub and regrowth	Non-woody vegetation	
	Sedge swamp (non-woody vegetation)	
Burnt vegetation	Recently burned area (no vegetation)	
Cultivated and managed land	Cultivated land	
	Water	Blackwater lake
		Rivers

This was necessary because originally the TREES classification was based on SPOT (Satellite Pour l'Observation de la Terre) vegetation imagery with a spatial resolution of 1 km, whereas the classification in this study was conducted using finer-resolution (30 m) Landsat data. The land-cover legend was divided into two main categories, namely forest and non-forest vegetation cover (Table 1).

Secondary vegetation, developed after burning, was represented by three land-cover classes consisting of a sequence from the most to the least advanced classes of post-fire vegetation regrowth, namely (i) secondary forest, (ii) mosaic of trees and non-woody vegetation, and (iii) non-woody vegetation, dominated by ferns. Discrimination between secondary forest and mosaic of trees and non-woody vegetation was based on the proportion of tree canopy cover to non-woody vegetation. Scattered patches of trees were classified as mosaic of trees and non-woody vegetation, whereas woody vegetation with a closed canopy was classified as secondary forest. In total, 15 land-cover classes were distinguished and land-cover maps were derived for 6 years, 1973, 1991 (before the dry season, i.e. before fires started), 1993, 1997 (before fires started), 2000 and 2005.

Burned area detection

Detection of burned area requires either pre- and post-fire images or a single post-fire image taken not long after the fire event. The active fire data, obtained almost in real time by the spectroradiometer MODIS, provided by the Web Fire Mapper, were used to ascertain the length and time-range of the burning season for the fires in 2002, 2004 and 2005 following the approach of Li *et al*. (2000). This was important because we needed to obtain information on the length and timing of the burning season in each year under investigation in order to avoid inadequate data purchase and analysis.

Burned area detection was undertaken using both the traditional method, based on manual interpretation, and a digital automatic method. The former was applied only to the archival data obtained before 2000; the automatic detection of burned areas on these images was limited by rather poor-quality imagery (haze contamination). The burn scars from the 1973, 1991 and 1997 fires were, therefore, interpreted manually based on single post-fire Landsat MSS and TM scenes. Visual interpretation in conjunction with land-cover mapping was used to derive the burn scars resulting from fires during the period 1974–90.

After the 1997 fire, the pattern of burned areas had a more patchy character, with a large number of scattered, smaller burn scars that were difficult to delineate manually, thus a more robust, automatic method was applied. The burned areas from the 2002 fire were delineated using bi-temporal differenced Normalised Burn Ratio (dNBR) index values derived from observations of the surface made at infrared ($IR = 0.86\,\mu m$) and short-infrared ($SWIR = 2.2\,\mu m$) wavelengths calculated from pre- and post-fire images recorded by the Landsat sensor (Eqn 1). This technique has been used effectively to detect burned area in another tropical peatland area in South-east Asia (Phua *et al*. 2007) and in other ecosystems (Epting *et al*. 2005; Loboda *et al*. 2007). The fires of 2004 and 2005 could not be mapped using Landsat imagery owing to a data availability problem; therefore, a time-series of Disaster Monitoring Constellation (DMC) images was used as a substitute. The NBR index could not be calculated from the DMC data because of the absence of the SWIR band, which is required to compute the NBR. Thus the masks of burned area for 2004 and 2005 were derived from a time-series of 20-day image composites constructed using a maximum Normalised Difference Vegetation Index (NDVI) value criterion. NDVI was calculated from the red ($R = 0.68\,\mu m$) and infrared ($IR = 0.86\,\mu m$) wavelengths (Eqn 2). This index has been widely used to detect burned areas in temperate and boreal forests (Li *et al*. 2000; Chuvieco *et al*.

Table 2. Land-cover statistics for Block C of the former Mega Rice Project (MRP) for the period 1973–2005

Land-cover data for 1991 and 1997 were obtained from images acquired before fires, thus the considerable land-cover changes that occurred as a result of the 1991 and 1997 fires are reflected in the figures for 1993 and 2000 respectively. Percentage values are relative to Block C (448 925 ha)

Land-cover type	1973		1991		1993		1997		2000		2005	
	(ha)	(%)	(ha)	(%)	(ha)	(%)	(ha)	(%)	(ha)	(%)	(ha)	(%)
Mixed peat swamp forest	253 598	56.5	215 841	48.1	182 255	40.6	177 626	39.6	72 027	16.0	53 632	11.9
Low pole peat swamp forest	14 135	3.1	14 135	3.1	14 135	3.1	14 135	3.1	2934	0.7	0	0.0
Heath forest	10 929	2.4	14 343	3.2	14 343	3.2	14 343	3.2	14 343	3.2	13 736	3.1
Mangrove forest	9276	2.1	9276	2.1	9276	2.1	9276	2.1	9276	2.1	5979	1.3
Freshwater swamp forest	39 100	8.7	37 912	8.4	35 314	7.9	34 991	7.8	26 898	6.0	26 640	5.9
Fragmented and degraded mangrove	9174	2.0	9174	2.0	9174	2.0	9174	2.0	9174	2.0	10 546	2.3
Secondary forest	0	0.0	2677	0.6	2677	0.6	4617	1.0	1615	0.4	18 262	4.1
Non-woody vegetation	36 598	8.2	59 243	13.2	59 171	13.2	85 181	19.0	200 792	44.7	160 145	35.7[A]
Sedge swamp	9609	2.1	9609	2.1	9609	2.1	9609	2.1	9609	2.1	8267	1.8
Mosaic of trees and non-woody vegetation	10 038	2.2	18 832	4.2	18 905	4.2	25 271	5.6	40 329	9.0	48 843	10.9
Heavily logged peat swamp forest	0	0.0	0	0.0	1940	0.4	0	0.0	0	0.0	0	0.0
Recently burned area	31 109	6.9	0	0.0	33 198	7.4	4015	0.9	0	0.0	55 349	12.3
Cultivated land	16 617	3.7	49 140	10.9	50 184	11.2	51 942	11.6	53 184	11.8	43 907	9.8
Blackwater lake	4169	0.9	4169	0.9	4169	0.9	4169	0.9	4169	0.9	0	0.0
Water body	4573	1.0	4573	1.0	4573	1.0	4573	1.0	4573	1.0	3619	0.8

[A]In 2005, the decrease in the area of non-woody vegetation was caused by the fact that 9.7% of the study area occupied by this class was classified as recently burned area; the area of cultivated land also declined for the same reason.

2002). The 2004 burn scars were derived from a maximum NDVI composition obtained at the beginning of the 2005 dry season (i.e. just before the fire season started), whereas the burn scars of 2005 were estimated from a post-fire (end of dry season) maximum NDVI composite.

$$\mathrm{NBRpre} = (\mathrm{IR} - \mathrm{SWIR})/(\mathrm{IR} + \mathrm{SWIR})$$

$$\mathrm{NBRpost} = (\mathrm{IR} - \mathrm{SWIR})/(\mathrm{IR} + \mathrm{SWIR})$$

$$\mathrm{dNBR} = \mathrm{NBRpre} - \mathrm{NBRpost} \tag{1}$$

where IR = 0.76–0.9 μm; and SWIR = 2.08–2.35 μm.

$$\mathrm{NDVI} = (\mathrm{IR} - \mathrm{R})/(\mathrm{IR} + \mathrm{R}) \tag{2}$$

where R = 0.63–0.69 μm; and IR = 0.76–0.9 μm.

The masks of burned areas for the 2002, 2004 and 2005 fires were obtained by setting a threshold for the dNBR or NDVI values at an empirically determined level determined from the frequency distribution of index values over sample test sites (Loboda *et al*. 2007).

Results

Land cover 1973–2005

In 1973, just after the fires that occurred in that year, forest covered 73% (327 038 ha) of the total area of Block C, of which 60% was defined as PSF (57% mixed-PSF and 3% low-PSF) (Table 2 and Fig. 2). Other forest types (heath, mangrove and freshwater swamp forest) occupied 13%. The remaining 27% of the total land area comprised mainly recently burned areas (7%), non-woody vegetation (8%), cultivated land (4%), and mosaics of trees and non-woody vegetation (2%) (Table 2). Eighteen years later, in 1991, just before the fires of that year commenced, forest still dominated a large part of the study area (65%), with 51% occupied by PSF. The rest of the land was covered mainly by non-woody vegetation (13%), cultivated land (11%) and mosaic of trees and non-woody vegetation (4%). Two years later, in 1993, forest cover had decreased by 8%, now accounting for 57% of the study area, with PSF occupying 44% of the land cover.

In 1997, just before that year's fire season started, forest still covered more than a half of the study area, of which 43% was PSF and 13% other forest types. Non-woody vegetation was the second most dominant land-cover type, accounting for 19% of the entire area, following by cultivated land (12%), mosaic of trees and non-woody vegetation (6%), sedge swamp (2%) and degraded mangrove (2%). An analysis of the land cover for the year 2000 (i.e. after the 1997 fires) demonstrated major changes in the land-cover pattern and showed for the first time the dominance of non-woody vegetation over forest. Non-woody vegetation occupied ~45% of the study area at that time, whereas forest covered only 28%, of which 16% was mixed-PSF and only 0.7% low-PSF. The other forest types occupied the remaining 11%, but with a noticeable reduction in freshwater swamp forest to 6%. The remainder was occupied mainly by cultivated land (12%), and mosaics of trees and non-woody vegetation (9%).

This trend in land-cover change continued over the subsequent 5 years. By 2005, 36% of the study area was dominated by non-woody vegetation, less than in 2000 because 10% of the study area occupied by non-woody vegetation was burnt in the 2005 fire and thus was classified as recently burned. The forest cover was reduced to 22%, including PSF (12%), freshwater swamp forest (6%), heath forest (3%) and mangrove forest (1%). Remaining patches of low-PSF and the area occupied previously

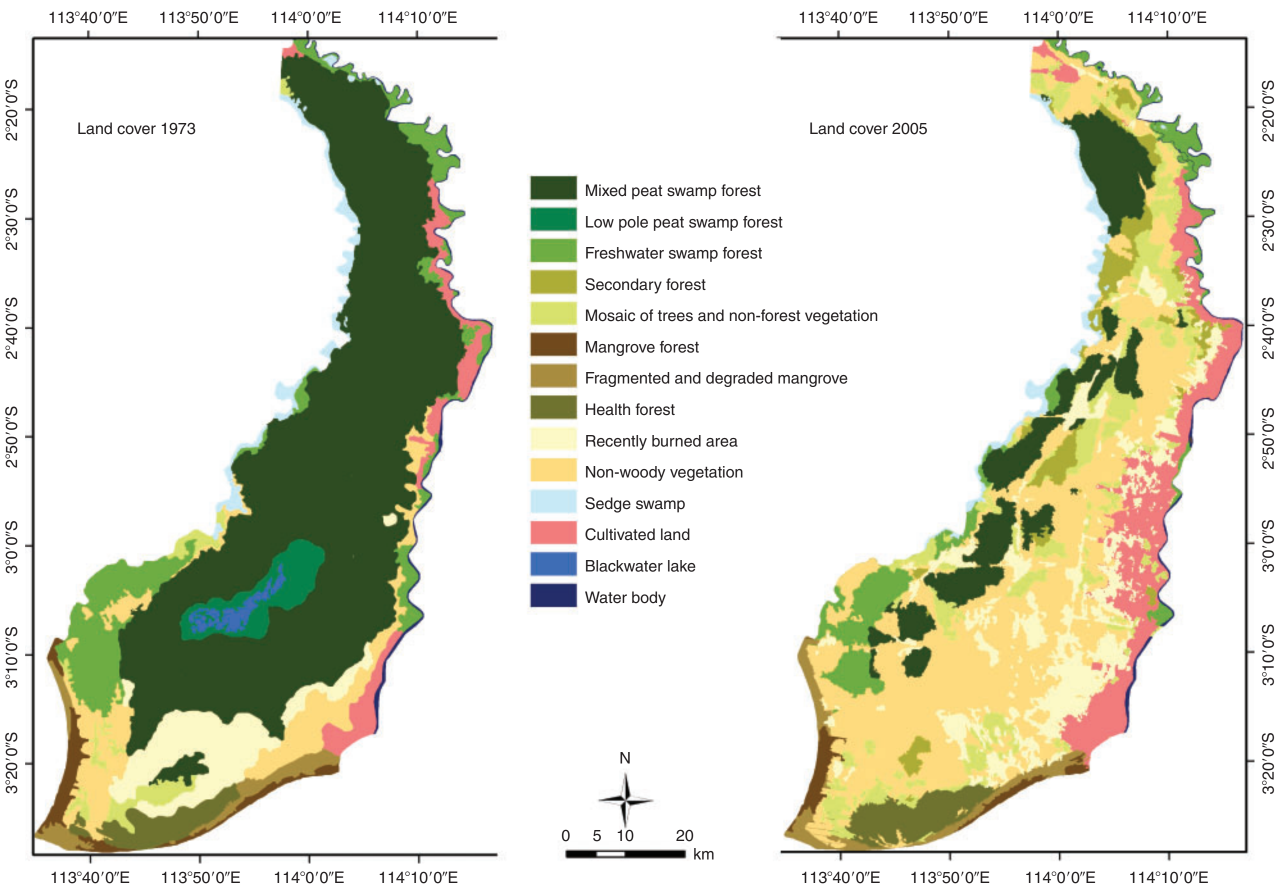

Fig. 2. Land cover of Block C, former Mega Rice Project, Central Kalimantan, Indonesia in 1973 (left) and 2005 (right).

by blackwater lakes on the central part of the peatland dome were now converted to non-woody vegetation. The forest loss was compensated by an increase in secondary regrowth vegetation comprising both mosaic of trees and non-woody vegetation (11%) and secondary forest (4%).

Fire extent and fire-affected area by land-cover type

In 1973, ~7% of Block C was affected by fire (El Niño drought conditions prevailed in the area in 1972–73 (Wyrtki 1975; Aiken 2004)). Between 1973 and early 1991 (before that year's dry season started), ~9% (38 420 ha) of the study area was burnt. Approximately 98% of the total burned area at that time was in mixed-PSF, with the remaining 2% in freshwater swamp forest (Table 3). During the 1991 El Niño-related dry season, fire destroyed an additional 8% of Block C, mainly PSF. More extensive forest loss occurred during the 1997 fire that was associated with a very strong El Niño drought; fire affected ~34% of Block C (150 486 ha) at that time, with most fires located in PSF including both mixed- and low-PSF. Of the total area burnt in 1997, 79% was in PSF (~72% mixed-PSF and ~8% low-PSF), and 6% was in freshwater swamp forest. Subsequent fires during the 2002 El Niño drought affected more than 22% of Block C (99 573 ha). This time the most fire-affected land-cover class was non-woody vegetation (65% of total burned area), followed by PSF (17%) and mosaic of trees and non-woody vegetation (5%); cultivated land also contributed to the total burned area (7%).

During the following dry seasons of 2004 and 2005 (both non-El Niño years), fire destroyed 14% (64 562 ha) and 12% (55 349 ha) of the study area respectively, occurring mainly in non-woody vegetation. The 2004 fire affected 6% of the remaining PSF (5% mixed-PSF and 1% low-PSF) while the area formerly occupied by the blackwater lake system was burnt for the third time. Of the total area burnt during 2004, 72% was non-woody vegetation and 11% was cultivated land. A similar pattern of fire occurrence took place during 2005, when overall 79% of the total burned area was occupied by non-woody vegetation, 16% by cultivated land, and 5% by mosaic of trees and non-woody vegetation, with just 0.2% accounted for by mixed-PSF.

Fire frequency

Comparison of the pre- and post-MRP periods (1973–96 and 1997–2005) reveals a distinctive variation in the fire frequency pattern, with an evident increase in the number of reburned locations through the later period. During the post-MRP period, more than half (54%) of the total study area was burnt at least once. Of the total burned area over that time (1997–2005),

Table 3. Total burned areas, proportion (%A) of fire-affected areas by land-cover type and proportion (%B) of total land-cover type affected by each fire in Block C (448 925 ha) of the former Mega Rice Project (MRP)

Land-cover type	1973	1974–90		1991		1997		2002		2004		2005	
Total burned area (percentage of Block C)	6.9	8.6		7.7		33.5		22.2		14.3		12.4	
Total burned area (ha)	31 109	38 420		34 413		150 486		99 573		64 562		55 349	
Fire-affected areas by land-cover type		(%A)	(%B)	(%A)	(%B)	(%A)	(%B)	(%A)	(%B)	(%A)	(%B)	(%A)	(%B)
Mixed peat swamp forest		98.3	14.9	92.0	14.7	71.5	60.5	14.9	20.5	5.3	6.0	0.2	0.2
Low pole peat swamp forest						7.5	80.1	1.8	61.3	0.8	100		
Heath forest								1.1	7.3			0.1	0.3
Mangrove forest								0.1	0.8			0.1	0.9
Freshwater swamp forest		1.7	3.0	7.5	6.9	5.8	23.1	1.0	3.7	2.5	6.3		
Fragmented and degraded mangrove								0.3	3.3			0.2	1.2
Secondary forest							20.9	0.3	16.6	0.2	11.4	0.3	0.8
Non-woody vegetation						5.1	9.1	65.1	29.8	72.3	23.0	78.6	21.3
Sedge swamp						2.5	40.0	1.7	0.8	0.7	4.4		
Mosaic of trees and non-woody vegetation				0.5	0.9	1.6	9.6	5.0	24.8	5.7	12.2	4.5	4.8
Cultivated land						3.2	9.3	6.8	12.7	11.4	13.8	16.0	16.8
Blackwater lake						2.7	100	2.1	50.8	1.1	100		
Non-peat swamp forest burnt		1.7		8.0		21.0		83.4		93.9		99.7	
Peat swamp forest burnt		98.3		92.0		79.0		16.7		6.1		0.2	

Table 4. Land-cover change over a 32-year period 1973–2005 in Block C of the former Mega Rice Project (MRP)

Rates are calculated based on two observations, one at the beginning and one at the end of the study period and then divided by the number of years between observations; negative values indicate loss; 1973–96 refers to the pre-MRP period, whereas 1997–2005 values are for the post-MRP period; Δha, total area of changes (in hectares) during the period 1973–2005

Land-cover type	1973–96		1997–2005		1973–2005		
	(% year^{-1})	(ha year^{-1})	(% year^{-1})	(ha year^{-1})	(% year^{-1})	(ha year^{-1})	(Δha)
Mixed peat swamp forest	−1.2	−3165	−8.7	−15 484	−2.5	−6245	−199 966
Low pole peat swamp forest	0.0	0	−12.5	−1767	−3.1	−442	−14 135
Heath forest	1.3	142	−0.5	−70	0.8	89	2807
Mangrove forest	0.0	0	−4.4	−406	−1.1	−101	−3297
Freshwater swamp forest	−0.4	−171	−3.0	−1042	−1.0	−389	−12 459
Fragmented and degraded mangrove	0.0	0	2.0	187	0.5	47	1372
Secondary forest	0.0	192	37.3	1724	0.0	575	18 262
Non-woody vegetation	5.5	2024	17.4	14 803	14.3	5219	123 547
Sedge swamp	0.0	0	−1.7	−168	−0.4	−42	−1342
Mosaic of trees and non-woody vegetation	6.3	635	12.9	3257	12.9	1290	38 805
Cultivated land	8.9	1472	0.2	104	6.8	1130	27 290
Blackwater lake	0.0	0	−12.5	−521	−3.1	−130	−4169
Peat swamp forest (mixed- and low-PSF)	−1.2	−3165	−9.0	−17 251	−2.5	−6687	−214 101
Non-peat swamp forest	−0.1	−29	−2.6	−1518	−0.7	−401	−12 949
Forested area	−1.0	−3194	−7.5	−18 768	−2.2	−7088	−227 050

55% was affected by single fire, 37% burnt twice and 8% burnt three or more times. In comparison, analysis of the distribution of burn scars for 1973–96 reveals that most fires occurred in new locations (99% of the burned area was affected by single fire), indicating a very low incidence of repeat fires.

Discussion

In South-east Asia, PSF has been under tremendous pressure from unsustainable land and forest management over recent decades (Rieley and Page 2005; Hooijer *et al.* 2006). In the study area, there has been an annual reduction in forest cover of ~2.2%, equivalent to ~7000 ha year^{-1} (in total ~227 000 ha of forest) over the 32 years covered by this investigation, with PSF experiencing the highest rate of loss of 2.5% year^{-1} (6687 ha year^{-1}, in total ~214 100 ha) (Table 4). A similarly high annual deforestation rate of 2.2% was reported by Hooijer *et al.* (2006) for PSF in Central Kalimantan during the period 1985–2000. The change in land cover from 1973 to 2005 is non-linear and two distinct phases can be identified, with major changes taking place from 1997 onwards following the start of the MRP. During the pre-MRP period (1973–96), forest cover changes were relatively minor and gradual (decline at an average rate of 1.0% $year^{-1}$ in forested area and 1.2% $year^{-1}$ in PSF). By 1991, almost half of the deforested land had been converted into cultivated land allied to transmigration programs in the south-eastern part of the study area (Fearnside 1997). Acceleration of the rate of land-cover change, with a rapid increase in the rate of forest loss, occurred following the commencement of the MRP infrastructure development to create rice fields. Over the 9 years 1997–2005, forest cover declined at a rate of 7.5% $year^{-1}$, with PSF being lost at a rate of 9% $year^{-1}$ (17 251 ha $year^{-1}$). For the first time, fires affected areas of overdrained low-PSF and peatland occupied formerly by blackwater lakes, resulting in the complete, irreversible loss of these two land-cover classes. In a predisturbance condition, these features would have occupied the wettest, highest parts of the peat dome (Page *et al*. 1999), but the intensive drainage caused by the MRP canal system lowered the watertable in the peat, thereby increasing the risk of fire (Dennis *et al*. 2005; Wösten *et al*. 2008).

Fire occurrence depends on quality and quantity of available fuel and presence of ignition sources; thus, fire occurs and propagates in fire-prone locations with sufficient fuel to maintain fire. The remote location of the extensive burn scars recorded on the 1973 image (Fig. 2) in the southern part of the study area may suggest a natural source of ignition because the fire occurred some considerable distance from habitation. Ignition would be more likely to occur if fire had either been preceded by a severe period of drought or if lightning struck a fire-prone surface. In the case of this particular fire, both factors could have been important. First, 1973 was an El Niño year (Wyrtki 1975; Aiken 2004), and the fire location was possibly covered by fire-prone vegetation. This hypothesis was based on the structure of the regenerating vegetation that, even three decades after burning, showed no progression to forest, suggesting that the area had been burnt more than once before the 1973 fire. Lightning-ignited fires have been noted in the past in PSF (Brünig 1973). In addition, a sharp and explicit border excluding the burn scar from the adjacent PSF confirmed the resistance of the undisturbed forest to fire (i.e. the 1973 fire). The high humidity of intact PSF acts as a fire suppression barrier, protecting both the forest vegetation and the peat surface from burning (Siegert and Ruecker 2000; Wösten *et al*. 2008).

Other fires within the region in the post-1973 period appear to have been strongly linked to human access. Between 1973 and 1996, most burn scars were located along forest edges (i.e. in disturbed forest) and usually in close proximity to human settlements. These were probably related to low-intensity slash-and-burn traditional land-use practices in which natural resources were obtained from forest areas near to rivers (limited access) while some forest was converted to temporary agricultural land for the growth of subsistence crops (MacKinnon *et al*. 1996; Rieley and Page 2005). Intact PSF was also the target of labour-intensive logging (Siegert *et al*. 2001). By 1991, a network of log-extraction tracks covered almost the entire area of mixed-PSF, but avoided the less commercially valuable low-PSF. In spite of these practices, PSF remained relatively unaffected by fire, even during the extended droughts associated with the El Niño events of 1973, 1982 and 1991, because under relatively undrained conditions, the watertable is naturally close to or above the peat surface for much of the year. From 1997 onwards, however, the natural resilience of the peatland landscape was changed dramatically. Intensive drainage impaired the peatland hydrological system, contributing to the desiccation of the peat, thus increasing the risk of combustion (Rieley and Page 2005; Hooijer *et al*. 2006). Consequently, both the vegetation and the peat surface became drier, adding to the total amount of fuel available to burn, and intensive surface fires supported ground fires that were extremely hazardous and difficult to extinguish (Usup *et al*. 2004). In addition, the canals allowed easy access for people into previously remote areas of peatland and their activities provided ignition sources; thus, many of the fires in 1997 and subsequent years were located close to canals. The widespread, intensive fires of 1997, associated with a strong El Niño drought, initiated a period of enhanced fire activity in the study area. In 1997 alone, fires damaged over 33% of the study area, 10% more than had burnt during the previous 23 years. Five years later, during a weak El Niño in 2002, fires affected 22% of the study area. In 2004 and 2005, fires during 'normal' dry seasons (i.e. independent of El Niño-induced drought conditions) burned 14 and 12% of the study area respectively. The study area was also severely affected by fires in 2006 (Field and Shen 2008; van der Werf *et al*. 2008) and in 2009, associated again with the El Niño phase, indicating the continuing vulnerability of the deforested and drained peat swamp ecosystem to recurrent fires.

Most of the fire-affected, deforested peatland in Block C of the former MRP has become dominated either by non-woody vegetation dominated by fire-prone ferns or a mosaic of trees and non-woody vegetation, but it has not been converted to agriculture. These exposed, abandoned peatlands are now subject to intensive peat loss and release of carbon through both combustion of vegetation and peat (Page *et al*. 2002; Ballhorn *et al*. 2009) and accelerated peat decomposition (Jauhiainen *et al*. 2008; Hirano *et al*. 2009) and are contributing to high regional carbon emissions. Hooijer *et al*. (2006), for example, calculated that on average 0.6 Gt of CO_2 $year^{-1}$ is emitted annually to the atmosphere from South-east Asian peatlands through peat oxidation.

This study provides strong evidence that fire is now an important factor influencing the former MRP peatland landscape, where many locations are experiencing repeat fires. Of the total burned area between 1997 and 2005, ~45% was subject to multiple fires, with 37% burnt twice and 8% burnt three and more times. The increasing trend in fire occurrence has been also observed on a larger scale by Langner and Siegert (2009), who demonstrated using hotspot data that 6.1% of forest in Borneo had been affected by fire more than once over the period 1997–2006. These results suggest a positive feedback between deforestation, drainage, fire susceptibility, fuel loading and fire severity that has been recognised in several tropical ecosystems (Cochrane 2003; Cochrane and Barber 2009). This, in part, reflects the fact that once burned, forest is more fire-prone owing to the high amount of dry and flammable materials left over from the previous fire and the lower humidity that results from a greatly reduced tree canopy (Uhl and Kauffman 1990; Cochrane and Schulze 1999; Siegert *et al*. 2001). Not all potential fuel burns in each fire event. This is particularly the case for initial fire when tree leaves and branches and surface non-tree vegetation burn but most tree trunks remain unburnt as fuel for subsequent fires. Many trees in forested peatlands also fall to the ground because of loss of peat soil whereas some remain standing but dead. This positive feedback, however, seems to be disrupted in this peatland area after a few consecutive fires, because after a second intensive fire, most of the aboveground vegetation is eliminated and more time is required before the vegetation can rebuild a sufficient amount of biomass and hence fuel load to sustain a new fire. Until 1997, PSF tree biomass was the main source of fire fuel, but following the extensive fires that occurred during that year, this changed to non-woody vegetation, dominated by ferns. For example, 70% of the fire-affected area during the 2004 and 2005 fires was dominated by non-woody vegetation whereas 6% consisted of mosaics of trees and non-woody vegetation. This change in fuel type has probably influenced the fire regime, through changes in intensity and type of combustion, favouring more surface fires rather than ground, i.e. deep peat fires, which require a higher fuel load and fire temperature before they can become established but then smoulder at a lower temperature for longer time (Usup *et al*. 2004; Page *et al*. 2009). This may support the negative fire feedback with an annual burning regime that was observed by Balch *et al*. (2008) in a transitional forest in the south-eastern Amazon.

Despite increased fire activity observed since 1997, some locations within the study area show regeneration potential. Mosaics of trees and non-woody vegetation, indicative of an initial stage of forest regeneration, increased from 6% in 1997 to 11% in 2005, at a rate of 13% year^{-1}, while secondary forest increased by 4% over the same time period. This secondary vegetation is fragmented but if fire is suppressed and fragments link up to cover larger areas, they have the potential to develop naturally to secondary peat swamp forest. Further research is being undertaken to investigate the long-lasting effects of multiple fires on vegetation recovery and to assess the regeneration capacity of post-fire vegetation in order to predict the direction of vegetation succession and devise appropriate restoration strategies where natural succession is halted.

It is also important to understand that protecting the remaining peat swamp forests from logging, drainage and land conversion should be a high priority as undisturbed forests are at very low risk of combustion whereas, as this study has illustrated, degraded peat landscapes enter a downward spiral of ecological change driven by increasingly frequent fires.

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